Promoting Successful Restoration through Effective Monitoring in the Chesapeake Bay Watershed

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STREAMS

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Introduction and Acknowledgements

The National Fish and Wildlife Foundation (NFWF) funds a variety of projects aimed at achieving restoration goals for the Chesapeake Bay and watershed. In order to use their restoration funds most cost-effectively, NFWF seeks scientifically-based criteria for identifying projects with the highest potential for success and monitoring successful outcomes. This chapter is one of many being developed to generate monitoring metrics for restoration and management projects that include: 1) tidal wetlands; 2) non---tidal wetlands; 3) streams; 4) “green” stormwater management (under the direction of Allen Davis, UMCP); 5) agricultural and forest management (under the direction of Brian Benham and Gene Yagow, VA Tech); and 6) stewardship and social marketing (under the direction of Gene Yagow and Erin Ling, VA Tech).

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Stream Restoration - Implementation Verification, Monitoring, and Adaptive Management

This document provides a summary of evidence-based metrics for verifying project implementation and outcomes from stream restoration. Since most restoration is at the reach scale but stream recovery is dependent on upstream and watershed conditions, we also identify major constraints on restoration which are critical to evaluating restoration success. The metrics selected were intended to fulfill the goals of demonstrating potential benefits from restoration activities, promoting adaptive management, and enhancing understanding of effective restoration techniques. Metrics have been selected on the basis of expert guidance and a literature review that identified metrics that captured meaningful ecological or environmental outcomes. A *meaningful* outcome was defined as the restoration of desirable ecosystem processes and functions (e.g., nutrient cycling or sediment accretion), which are considered some of the best indicators of a system's ability to produce beneficial stream properties (e.g., enhanced fish habitat, water quality improvements) and to sustain them into the future. We are not implying that the metrics we recommend directly represent ecological functioning (i.e., they are not surrogates), but these are the best metrics given that the time frame and funds for restoration monitoring of NFWF projects may be limited. The metrics require a range of measurement approaches from the simple to the specialized. Therefore, it is not expected that all restoration projects would implement the full range of metrics.

This report is organized as follows:

1) Project Types
2) Discussion of links between metric selection and project goals
3) Discussion of implementation verification metrics
4) Discussion of class of monitoring metrics used to support functional outcomes and goals
5) Tables with suggested monitoring metrics and a prioritized list
    - Table 4. Stream restoration monitoring metrics
    - Table 5. Prioritized sets of recommended metrics
6) Appendix A: Scientific support for monitoring metrics

Project Types

Streams and rivers link upland processes and downstream receiving waters and thus play an important role in determining the overall health of the Chesapeake Bay. The major goals and associated approaches for stream restoration projects implemented in the Chesapeake Bay watershed have been extensively reviewed by Hassett et al. (2005) and can be classified into six major types of projects:

riparian management - The replanting of native riparian vegetation and/or the removal of non-native plant species.

bank stabilization - Practices designed to reduce excessive erosion or slumping of stream banks.

water quality management - Practices that aim to protect or restore water quality (e.g., livestock exclusion).

in-stream habitat improvement - Practices designed to increase habitat availability and provide refugia for in-stream organisms (e.g. installation of habitat structures).

channel reconfiguration and floodplain reconnection - Engineering approaches to mimic natural conditions (e.g. alteration of sinuosity) or improve connections with the floodplain (e.g. bank grading).

dam removal - Removal of in-stream features that serve as barriers to fish passage. Usually includes elements designed to enhance bank stability upstream of the removal site

Metric Selection to Support Project Goals

Choosing and interpreting an appropriate set of monitoring metrics will depend on project goals. Stream restoration projects often have multiple goals, but primary goals often include minimizing erosion and channel down-cutting or providing habitat benefits for species that are considered iconic of healthy streams (e.g., mayflies, stoneflies, and caddisflies). Other goals may include improving water quality especially if the stream is a tributary close to its receiving lake or coastal area, or, restoring fish that are highly valued (e.g., salmonids) or threatened or endangered (e.g., mussels, sturgeon) . Improving aesthetics and enhancing access for recreational use are also common goals for stream restoration. To promote these goals, numerous environmental processes must be restored, including hydrologic, geomorphic, and ecological processes.

Often, stream restoration success towards multiple goals is judged in terms of whether the project achieves a self-sustaining state that over time comes to closely resemble a natural condition in terms of structure and function (Palmer et al. 2005). However, position in the watershed, landscape context, and other factors such as water extractions for irrigation can limit what is achievable. The reality is that there is often a need to balance goals with site constraints. In some cases projects may be designed to fulfill a narrower set of goals, such as sediment retention or removal of non-natives, in a location where full restoration of natural dynamics and habitat is not possible. These projects can still be considered successful for a narrow set of goals.

Another consideration when setting restoration goals is that goals can conflict with one another and in such cases, metric selection and interpretation will need to be adjusted to reflect priorities. For example, stabilizing streambanks to minimize their erosion may make it impossible to allow floodplain over-flow to increase habitat for species whose life histories depend on such habitat for spawning or foraging. There are also many constraints that operate because of anthropogenic factors that cannot be reversed (e.g., nearby infrastructure may require protection from flooding or high levels of imperviousness upstream of a project site may preclude restoration of a natural flow regime). To manage conflicts between the desire to maximize ecological restoration and meet human needs, many stream restoration practitioners have increasingly moved to restoration approaches that combine novel designs to accomplish as much as possible given the constraints and conflicting uses (Filoso and Palmer 2011). Finally, because it may take a long time for a restored stream to reach a point in which the structure and function are similar to unimpacted streams, the challenge is to identify indicators of progress. Indicators of progress should be measurable, detectable within the first few years following project completion, and should indicate whether a rehabilitated river reach is heading towards the condition of good ecological status (Mathews et al. 2010).

The literature review was used to link metrics to the functional outputs that they measure (Table 1) in order to promote selection of metrics that match general goals such as habitat for characteristic species assemblages. However, for more specific goals, such as restoration of habitat for a specific species or to maximize a specific process (e.g., nitrogen retention), the project design and monitoring metrics will need to be tailored to those specific outcomes, usually with the help of subject area experts. Because system processes interact in complex ways, a suite of monitoring metrics used together is often the best approach for understanding progress towards goals and to evaluate any tradeoffs between goals.

Table 1. Selected monitoring metrics and correspondence to project goals (based on literature evidence). Note - the *constraint metrics* discussed in the text are not listed (see Tables 3 and 5).

Category	Metric	Nitrogen Retention & Removal	Phosphorus & Sediment Retention	Vital habitat
Hydrologic	Discharge (S)	X	X	X
Geomorphic	Substrate Particle Size			X
	% Fines on streambed		X	X
	Floodplain connectivity	X	X	
	Streambed armoring	X		X
	Channel Geometry			X
Biotic	Species list (invertebrates or fish)			X
	% Sensitive (invertebrates or fish)			X
	Fish size structure			X
Riparian Vegetation	% native vegetation intact	X	X	X
	Vegetation composition & size	X	X	X
Physico-chemical	Water chemistry	X	X	X
	Temperature			X
	Oxygen	X		X

Implementation Verification Metrics

The purpose of implementation verification metrics is to confirm that the proposed project was executed according to the planned design. Verification metrics will vary by type of project and may include both structural and non-structural elements (Table 2). Verification will generally involve comparing design drawings to field implementation and a checklist may be created, specific to the project design, to ensure all key design elements have been followed. Any purposeful deviations from designs should be explained and justified prior to verification of implementation.

Table 2. Implementation Verification Metrics for Streams

Metric Category	Metric	Description
Structural	Built features (during construction)	Step pools, floodplain re-connection, or other built elements should have photos or site inspections to document that design elements were followed by contractors and disturbance to the riparian zone was minimized
	Built features (post-construction)	Built elements match design in extent, placement, and type of material – should be reassessed after one flood season
	Topography	Site slope matches design
	Natural structures	Coir logs, woody debris additions, or other natural structural elements meet all design specifications
Non-Structural	Riparian plantings	Verify that specified planting was conducted through photo documentation or on-site inspection
	Vegetated and non-vegetated areas and basic channel form	Measure/estimate area of pools, riffles, riparian corridor width; Use of GIS, Google maps, Lidar or other appropriate technology is desirable for documenting pre vs. post presence and extent of non-structural features including basic channel form. Detailed channel morphometric surveys (e.g., x-section and longitudinal profiles) are not necessary unless channel stability is a project goal.

Monitoring metrics

The purpose of monitoring metrics is primarily to provide evidence that the ecological functions of the site have been restored and to suggest when adaptive management is needed to realize a project's full potential. Adaptive management may include site maintenance, repeating some restoration activities (e.g., replanting vegetation), or modifying design elements (e.g., removal or repair of in-stream structures that have failed or are causing unintended problems). The metrics were developed with the assumption that sites have been pre-screened to promote successful restoration (Palmer 2009). The importance of appropriate site selection for ensuring project success cannot be overemphasized, since site constraints (Table 3) will largely drive what is achievable. For example, landscape conditions, such as connections to upstream tributaries, will affect whether some types of biota are able to make use of the site (Kail and Hering 2009).

Because watershed-scale factors are the primary determinants of the overall health of a stream, the extent to which a degraded reach can recovery following a restoration action may be limited depending on land use, upstream or riparian conditions. Since most stream restoration projects are implemented at reach scales, metrics that can act to constrain project outcome should be part of the pre-restoration assessment protocol (Table 3). If the watershed-scale features represented by the metrics change post-restoration or the watershed scale features are directly addressed by the restoration action, then the watershed metrics should be reported along with the site-scale metrics during post-restoration assessment.

The guidance provided here is intended to be the basis of a more detailed sampling design that specifies appropriate analytical techniques and the frequency and areal extent of monitoring. The ability to demonstrate statistically significant results requires that data collection follows appropriate techniques such as collecting replicated random samples. Such detailed plans should be developed through consultation with local professionals who can adapt these recommendations to the goals, site conditions, and resource availability specific to the project.

Table 3. Common constraints (and associated metrics) that may prevent full or partial restoration.

Impervious cover (% Imperviousness in the watershed draining to the reach of interest)	Flow regime (timing, magnitude, and duration of high and low flows) is considered a master variable that determines habitat suitability for many stream species and controls rates and direction of key biogeochemical and biophysical processes. High levels of impervious cover lead to flashy flows in streams that may prevent recovery of species and underlying processes needed to support a healthy stream community. Impervious cover is also typically associated with reduced water quality including higher stream temperatures, contaminants, and in some cases higher suspended sediment loads.
Dams or flow diversions (# and size of dams or ~ % of flow impacted by diversion)	As above, flow regime is a master variable. Depending on the design, flow diversions that re-route water reduce or accelerate local flows that may negatively impact geomorphic processes and in-stream habitat suitability . Dams of sufficient size with deep reservoirs significantly alter downstream temperatures and also lead to sediment 'starvation' downstream that can result in loss of habitat and associated species.
Agricultural land use (% of watershed in agricultural use)	Unless highly effective best management practices are in place (e.g., riparian buffers, winter cover crops), farmland adjacent to streams often leads to excessive nutrient and sediment levels in streams which prevents recovery of native species and alters some biogeochemical processes.
Invasive species of concern (presence of invasive species known to limit recovery, i.e. nuisance species)	Risks from invasive species vary by restoration goal. Invasive riparian plants (e.g., multi-flora rose) may limit restorability of diverse native plant and bird communities. Invasive aquatic species may: out-compete native species for habitat or food (particularly fish); may alter the streambank or stream bed in ways that impede recovery of some native species; may alter resource availability; or may prey on native species.
Source of colonists (present/absent)	Colonization by biota requires connectivity to established populations, which may be located upstream, downstream, or across land surfaces depending on the species of concern. If water quality above a reach is poor, then the likelihood of fauna re-colonizing a restored reach is dramatically reduced. In the case of fish, they may move from downstream reaches if there are no in-stream barriers to migration. If the restored reach is a headwater site, then colonization of macroinvertebrates primarily occurs when adult flying stages reach the site and oviposit eggs; however, adult flight generally occurs only through vegetated regions and then typically along the stream corridor. Thus lack of vegetation along the stream or between the reach and a nearby healthy stream will limit or preclude colonization.
Upstream water quality (temperature, oxygen, contaminants, etc.)	Poor water quality in the tributaries above a restoration site often prevents recovery of biota and reduction of sediment or nutrient flux. If polluted water is reaching a site, it does not matter how much structural restoration work is done, biological recovery will be unlikely. Similarly, large inputs of sediment or nitrogen upstream of a project may swamp any efforts to reduce the levels of suspended sediment and nutrient concentrations in a lower reach.

The monitoring metrics are intended to be used both before and after project implementation and may supplement data used to pre-screen locations. In addition, to properly interpret metric values at a site, **reference sites will be necessary** to 1) understand the characteristic spatial and temporal variability necessary to produce desired outcomes (White and Walker 1997) and 2) distinguish short-term fluctuations in environmental condition (e.g., drought, storms) from restoration effects. Reference sites may also be needed to generate metric expectations under regional drivers of change, such as land use or climate change. It is important to emphasize that reference sites are not morphological "template" sites that a restoration project is striving to become. All streams exhibit a range of physical and biological conditions even in their most pristine state. Therefore, the purpose of reference sites is to assess the range in "reference conditions" of biota (e.g., range in faunal diversity or abundance throughout all seasons; Palmer 2009), water quality (e.g., range of conditions across all seasons and flow levels; Poole et al. 2004), and channel morphology (i.e., the dynamic equilibrium concept).

Ideally, reference sites will be monitored simultaneously with project sites. However, if resources are not available to sample reference sites, it may be possible to use regional databases to put monitoring results in context. A great deal has been written about selecting reference sites for stream restoration and the concept has often been misused when it is viewed as a template for restoration (Palmer 2009). The important point to remember is that data from reference sites on channel form, biota, plants, and other characteristics are needed to understand the natural variability in least disturbed systems and provide some context for what may be possible (Falk et al. 2006) However, because many restoration sites are subject to some level of constraint (Table 3), reference data should be used to identify a desired trajectory for key system attributes that are inherently variable (e.g., ranges of flow and sediment inputs, variability in the location and number of habitat types, and changes in the species composition of assemblages through time and space; Hughes et al. 2005).

Collecting monitoring data across sites also helps in the development of standardized data sets that can be used to track cumulative restoration progress over time and test hypotheses regarding which aspects of location, design, and approach contribute to restoration success. To promote this goal, **basic project implementation information,** such as condition prior to restoration and restoration actions taken, **should be recorded**. Relevant information includes: latitude and longitude, stream order, land use and % impervious cover in the contributing watershed, extent of intact native riparian vegetation (buffer width and longitudinal extent), and constraints (e.g., upstream dams, infrastructure that limits options, etc).

The recommended monitoring metrics for stream restoration are shown in Table 4 and are broken down into four categories: Hydrology, Geomorphic, Biota, and Physico-Chemical, which require distinct monitoring approaches. The metrics were considered the most promising because the current literature orwell-accepted ecological theory suggests they are either *valid proxies for* or are *direct measurements of* functional outcomes. **Metrics are either structural, meaning that they quantify spatial conditions and patterns, or functional, meaning that they quantify dynamic processes.** An easy way to distinguish the two is that structural metrics can be meaningfully evaluated at a point in time (e.g., the number of EPT taxa - mayflies, stoneflies, and caddisflies - per unit length of stream or sampling effort), while functional metrics require multiple measurements over time to generate meaningful measures and thus have *rate* units (e.g., oxygen consumption per unit area per unit time). Functional metrics ensure that fluxes and dynamics (e.g., nutrient uptake) that influence biota are evaluated in relation to changing hydrologic conditions within a stream. Most of the metrics in Table 4 are structural and the hope is that they are proxies for functions, however, research to identify easily measured structural parameters that are true proxies for stream ecological functions is in its infancy (Palmer and Filoso 2009). Direct measurement of some ecological functions (e.g., denitrification, secondary production) are time-consuming and simple surrogate metrics are not yet available; however, measuring discharge (or some related hydrologic functional metric) remains essential because it is relatively simple to measure and provides a great deal of information about the system (as discussed below).

The table provides a brief description of the metrics, recommendations for measurement techniques, and some information about metric interpretation and use in adaptive management. These metrics are intended to be measured at multiple points along the restored stream through the use of transects or stratified sampling. For all metrics except the hydrology, replicate samples within each zone will be needed to do statistical analyses of change or to compare the restoration site to reference conditions. Sources of guidance for sampling techniques and measurement protocols are listed in the table.

Some of the metrics are useful in evaluating if the restoration project has been completely successful biologically (i.e., if the system is indistinguishable from reference sites). Although the presence of desired macroinvertebrates, fish, or the full suite of native biota indicates that the project was successful, when these bioassessment metrics fall short of the desired endpoint, they rarely provide insights into why the project was not successful. Thus, bioassessment metrics generally are not very useful for adaptive management unless there is detailed species-level information on the exact stressors limiting populations. Such information is rarely available, although there is ongoing effort to identify species whose loss is closely associated with specific stressors. In sum, bioassessment metrics reveal little about whether a restored site is on a trajectory toward recovery unless repeated measurements over very long time periods have been made and information on taxa-specific sensitivity to various stressors is available.

Hydrologic and Geomorphic metrics

Discharge and sediment dynamics influence virtually every aspect of stream ecological condition including two of the aspects of greatest interest to NFWF - reducing nutrient and sediment loads. The flow *regime* – magnitude, timing, and duration of high and low discharges – exerts control over ecological recovery following restoration. Ideally, restoration efforts should always seek to re-establish the historic flow regime because it directly influences biota and ecosystem processes (e.g., survival and reproduction), and because it exerts significant indirect effects through its impact on geomorphology, sediment transport, food availability, and habitat. However, because many streams are dammed or located in watersheds with highly altered land use, it is often impossible to recover the historic regime. Additionally, even if the regime could be re-established through some engineering, the sediment inputs in watersheds with dams or altered land use is fundamentally altered (often much lower inputs), meaning that restoration of the historic regime might result in flows too high given the sediment load (i.e., the flow has a certain sediment capacity which is not met by the incoming sediment load and therefore the channel erodes as the flow picks up sediment to meet its capacity this is known as sediment-starved flow).

Measuring discharge on a single or few dates provides little or no information on stream ecosystems or their response to restoration. However, there are fairly simple methods for measuring discharge over time and we stress that a **continuous record of discharge pre- and post-restoration is the most critical metric for a evaluating a restoration project.** With measurements of daily discharge, an annual hydrograph can be plotted (discharge vs date); however, multiple years of discharge measurement are needed to determine whether changes in the hydrograph are due to restoration or natural year-to-year variation in precipitation. A compromise is to make continuous discharge measurements (i.e., create annual hydrographs) at nearby reference sites during the same period measurements are taken at the restored sites. Any relative changes in discharge (pre vs post restoration) that did not occur at the reference site can then be considered as indicative (but not proof) of a restoration effect.

Relative changes in water level (" stage") pre vs. post restoration or in restored vs. control (unrestored) or reference sites (least-disturbed) can indicate changes in discharge and are easier to measure than discharge itself; however, stage-discharge relations (a "rating curve") are dependent on local channel conditions (channel width, depth, slope, and roughness). Since these local conditions often change pre vs. post restoration, it is essential that local stage discharge relationships be developed in order to obtain reliable discharge data. In sum, continuous discharge data for each of three years at the restoration site and a nearby reference site are the only sure way to determine how restoration influenced discharge.

The geomorphic parameters that are measured for a stream restoration project will vary somewhat depending on the goal. Since healthy stream channels are characterized by dynamic geomorphic adjustments, it can be difficult to determine when changes in channel geometry are significant (and thus of concern if, for example, keeping the channel in place was one of the project goals). If this is a concern, then basic channel assessments would include measuring channel width, depth, and incision or aggradation from cross-sectional or longitudinal surveys. Geomorphic parameters particularly relevant to biota include basic habitat features that different fauna prefer: pools, riffles, glides, etc. Finally, unless working in the Coastal Plain, measuring the amount of fine material (silts, clays, very fine sands) deposited on a streambed is useful in assessing habitat suitability for macroinvertebrates. If the project goal is to reduce the downstream flux of sediment, the 'sedimentation' of fine material in pools or in side channels or floodplains may be considered highly desirable, again illustrating that metric choice is strongly dependent on project goal.

For useful information and computer tools to assist in restoration designs as they relate to discharge, channel form, and sediment supply see the "Stream Restoration Toolbox" at the University of Minnesota's *National Center for Earth-Surface Dynamics (*http://www.nced.umn.edu/content/tools-and-data). There are also excellent texts on channel flow and sediment dynamics that may be useful to those new to this field (e.g., Gordon et al. 2004)

Biotic metrics

Assessment of the composition of benthic macroinvertebrates remains one of the most common forms of stream assessment. Benthic **c**ommunities are believed to provide an aggregate view of in-stream conditions because they respond to a variety of different stressors and integrate the impacts of those stressors over time (Barbour et al. 1996, Bonada et al. 2006). **While biotic assessments are good at evaluating the "status" of a stream ecologically, benthic and fish assessments are typically of little value in evaluating whether a restored stream is on a trajectory toward recovery for a variety of**

reasons. First, they are so sensitive to degradation that return of sensitive, "indicator" taxa typically requires almost full restoration of other ecological processes (e.g., flow, sediment, and temperature regime; food base). Second, biota may not be sensitive to the factors of most interest to Chesapeake Bay health**.** For example, they are insensitive to elevated nitrogen loads unless the N concentration is extremely high for a sustained period of time (Yuan 2010). Third, they generally provide little information that can be used for adaptive management if the biota have not returned. Biota respond to many stressors and may be unreliable as indicators of a specific problem, such as metal contaminants or high nutrient levels, because these pollutants are almost always associated with multiple, confounding effects. Consider, for example, that streams with high nutrient concentrations, such as those in agricultural areas, typically also have high levels of siltation and pesticides, making it difficult to use macroinvertebrate assemblages to pinpoint specific stressors.

Riparian Vegetation

Assuming there are no point source inputs of pollutants, the single most important factor for restoring a stream is to establish healthy, native riparian vegetation along the streambanks. There are good data on riparian composition, size, and structure for the entire Chesapeake Bay region (e.g., Rheinhardt et al. 1999; Rheinhardt et al. 2009.)

In a study of 30 Chesapeake Bay streams with riparian vegetation ranging from 0 to 50 years of age, Orzetti et al. (2010) found that habitat, water quality, and benthic macroinvertebrate metrics improved with age of restored buffer. Two aspects of the riparian vegetative cover can be used as metrics to measure restoration outcome. The percent of the stream corridor that is intact (**% Intact**) should be measured from the bottom of the restoration reach to the uppermost part of the headwaters. This is the most important factor influencing water quality, habitat, and stream temperature. If an individual project only restores the vegetation along a single "target" reach and there are gaps in the vegetated corridor above that reach then the project should report the % intact along the reach and the % intact above the reach; the second metric is a major constraining factor and may even render the proposed project reach unsuitable for restoration investments.

The composition of the vegetation (**% native vegetation**) and its spatial coverage (**% canopy cover)** relative to reference reaches are also important. Decreased canopy cover has been correlated with reductions in allochthonous inputs and benthic organic matter availability, which influence the food web and may determine species composition (Reid et al. 2008, Kaase and Katz in press). It is important to recognize that the composition and amount of riparian vegetation not only affects stream biota but also the abundance and diversity of birds, bats, spiders, and other terrestrial species For example, the number of tree species planted has been shown to influence the number of bird species (Gardali and Holmes 2011).

The fact that canopy cover can be highly correlated with basal food availability (algae and/or allochthonous inputs), water temperature, large woody debris, and biotic composition makes it a useful restoration metric. The goal should be to establish canopy cover similar to that in least disturbed reference sites. However, even if this is accomplished there may be many other important factors that limit restoration success. In particular, groundwater flow paths may circumvent the riparian root zone (e.g., naturally or because the stream is incised), and when this happens, high nutrient inputs from groundwater may lead to elevated stream nutrient concentrations despite an intact riparian corridor (e.g., Speiran 2010). In general, understanding flow paths to streams is critical to identifying priority riparian restoration sites and in understanding why nutrient levels in streams may be high despite vegetative cover (Weller et al. *in press).*

Physico-Chemical

Structural metrics such as those commonly used to evaluate habitat (e.g., substrate and physiochemical characteristics) may be misleading in terms of restoration success because they may not be the primary factor that led to stream impairment; often poor water quality due to excessive nutrients, sediments, or chemical contaminants must be addressed. Point measurements of temperature, pH, or other water quality parameters may be useful if pre-restoration levels of these metrics were seriously abnormal (e.g., exceeding thermal, pH or oxygen tolerances of native biota). However, more often there are small differences in these parameters compared to reference sites and frequent or continuous measurements are required. This is particularly true for assessing restoration of nutrient and sediment removal processes. **Direct measurement of water column nutrient and sediment concentrations coupled with discharge over the entire range of flows provides the best information.** Measurements taken only during baseflow or only during a single season would not be adequate to evaluate nutrient reduction benefits because movement of nutrients and sediments into streams peaks during storms and also varies dramatically with season and vegetative growth.

Table 4. Stream Restoration Metrics Table.

Category	Metric	Measured variables	Timing & Frequency (All measurements should be made pre and post restoration)	Main functions or processes supported	Performance Measures	Adaptive Management (AM)	Quantitative Assessments
Hydrologic	**Discharge**	- Volume of water passing a fixed point per unit time (e.g., m^3/sec; ft^3/sec) in the project reach - Access to nearest gage in the watershed? Incorporate context and spatiotemporal variability in discharge	- Continuous with a minimium of 1 year pre and post restoration (may be averaged daily or during storms, hourly) - Need a minimum of one water year of data - a few point in time measurements are not adequate	- Vital habitat - Nutrient and sediment removal	- Annual hydrograph more closely resembles pre-disturbanc or reference reach than prior to restoration (e.g., for urban restorations, peak storm flows attenuated) - If project designed to promote overbank flows, sediment deposition on floodplain will be evident -> move to geomorphic criteria	- Discharge may be 'adjusted' through dam management, channel adjustments, riparian planting	- Develop rating curve for stage vs. discharge (Relative changes may be evaluated pre vs. post using stage data)
Geomorphic	**Substrate Particle Size**	-Particle size distribution	-Semi-annually pre and post restoration	- Vital habitat	- Distribution of particle sizes on the bed more similar to reference stream. - Generally an increase in the diversity of particle sizes is considered desirable for biota (not true in Coastal Plain streams and not true for silts and clays) - Major changes over time may be indicative of project success, failure (depending on goals), or that there have been upstream changes that altered sediment supply to the project reach	- May add large bolders but generally AM can only be accomplished by restoring flow regime and sediment inputs	- Conduct 'pebble counts' along cross-sections and plot data (cumulative frequency graphs to calculate median and skew in particle sizes)
	% Fines	- Estimate fine particle coverage per unit area of streambed (fines = silts, clays, and small sand ~0.05 mm - 0.5 mm)	- Monthly or seasonally pre and post restoration	- Vital habitat	- % of streambed covered by fine particles more similar to reference stream. - Generally, less silts and clays covering the stream bed is considered beneficial for biota (not true for large rivers or streams with naturally fine sediments due to local geology)	- Evaluate flow regime to determine if peak flows are insufficient to prevent silt from accumulating on streambed - Evaluate sources of sediment in the watershed - Construction (if at very small scales) most often the source of excessive fine sediment	- Collect sediment samples; complete particle size analysis in lab to determine % of streambed particles < 0.5 mm
	Bed Armouring	- Surface D50:subsurface D50 (median particle size on streambed surface vs. a few centimeters below)	- Monthy to seasonally pre- and post restoration (this can be difficult to determine because the ratio may change during floods but not at baseflows when measurements are often made)	- Water quality - Vital habitat	-Ratio should decrease over time if streambed suffered from armoring prior to restoration	-Flow may not be sufficient to transport larger surface materials or sediment supply needs to be controlled	-Compare changes in the armor ratio over time
	Floodplain Accretion	- Rate of accumulation of sediments on floodplains across microhabitats, at various times	- Monthly to annually pre and post restoration	- Water quality	- Using photographs or visual measurements of fine sediment deposition near fixed features (e.g., tree trunks) - can be very difficult to do visually - Must determine that accumulated material is not mobilized and transported downstream during floods - May not be realized in a 3 yr period	- Flow regime needs to be evaluated in the context of channel geometry to determine if overbank flows and hydraulic retention in floodplain are sufficient to lead to accumulation	- Develop a sediment budget or use sophisticated methods for measuring accretion rates (e.g., radioisotopes, deployed sediment traps)

Table 4. Stream Restoration Metrics Table.

Category	Metric	Measured variables	Timing & Frequency (All measurements should be made pre and post restoration)	Main functions or processes supported	Performance Measures	Adaptive Management (AM)	Quantitative Assessments
	Floodplain connectivity	- Presence & extent of floodplain connected to stream at high flows	-Pre and post during moderate and high flows	- Water quality - Vital Habitat	- Using photographs and frequent visits, pressure transducers in the floodplain, or crest-gage	- Flow regime may need to be evaluated in the context of channel geometry if floodplain is rarely innundated. This may be due to channel down-cutting or lowering of the water table	- Stage recorder for which a rating curve is developed (depth of innundation over time) can be used to measure duration and frequency of floodplain innundation
	Channel Geometry	- Cross-section profiles (width:depth, bankfull depth, bank repose angle)	- Point in time measurements monthly to annually or more frequently during flood seasons; measure pre and post restoration	- Vital habitat	- No evidence of rapid and large-scale changes in stream morphology (note - streams are dynamic by nature - complete 'stability' is not desirable ecologically) - Measurements over time to determine if morphology has changed from 'as built' - May not observe significant change in 3 yrs	- Repair structures or make alterations to channel -requires a full hydrogeomorphic assessment to identify problems	- More detailed measurements may be able to detect smaller-scale changes in channel form (e.g., sophisticated survey equipment such as laser profilers)
		- Longitudinal profile (slope, riffle/pool size and profile, bar features) - % riffle-pool per reach	- Point in time measurements - Annually	- Vital habitat - Nutrient processing			
	Large/Coarse Woody Debris	- Presence of coarse/large woody debris	- Annually pre and post restoration	- Vital habitat - Nutrient & sediment retention	- Presence, count, and location - Ability to support macroinvertebrate and other taxa	- Large wood inputs can move during floods and need repositioning, particularly if they put infrastructure or other restoration elements at risk	- Number/reach length
Biota	**Species Lists (macroinvertebrates, fish)**	- List of taxa to highest resolution possible (e.g., species)	- Annually or seasonally pre and post restoration	- Water quality - Vital habitat	-Taxonomic composition includes a more diverse range of taxa (including sensitive species) than prior to restoration -Sensitive taxa colonizing reach indicates success -Approaches reference conditions -Observed vs. expected may be more indicative of performance trajectory	- It is very difficult to determine why particular taxa are absent using biotic metrics only; therefore, AM rarely employed based on this alone	- May use Surber samplers, electroshocking, or other sampling devices to gather organisms and determine taxa richness and composition per project reach(es)
	% Sensitive Taxa (macroinvertebrates, fish)	- % of individuals sampled that are are known to be sensitive to stream degradation (e.g., macroinvetebrates: mayflies, stoneflies, or caddisflies; salmonid fish)	- Annually or seasonally pre and post restoration	- Water quality - Vital habitat	- Taxonomic composition includes sensitive species that are found in least disturbed reference sites	- It is very difficult to determine why particular taxa are absent using biotic metrics only; therefore, AM rarely employed based on this this alone.	- Number of mayflies, stoneflies, and caddisflies down to species level - Abundance and total number of taxa
	Fish Size Structure	- Distribution of fish sizes	- Annually	- Water quality - Vital habitat	- Size distribution should include small individuals (1st year recruits) as an indication of successful reproduction	- It is very difficult to determine why reproduction is not occurring - thus other factors such as water quality and habitat must be evaluated for AM	- Formal fishery stock assessments include relating size to age, using statistics developed for estimating size of a fishery, etc.

Table 4. Stream Restoration Metrics Table.

Category	Metric	Measured variables	Timing & Frequency (All measurements should be made pre and post restoration)	Main functions or processes supported	Performance Measures	Adaptive Management (AM)	Quantitative Assessments
Riparian vegetation	**% Intact**	- % of stream (length) with intact riparian vegetation	- Annually pre and post restoration	- Water quality	- Increases over time	- May require replanting or protecting from deer grazing	- % of stream length that is fully vegetated
	Width	- Width of vegetative buffer	- Annually pre and post restoration	- Water quality	- Increases over time	- May require replanting or protecting from deer grazing	- Total width
	Composition	- % of plants native - Species list - % woody vs. shrub or grass	- Annually pre and post restoration	- Food web support	- More closely resembles pre-disturbance or reference reach after restoration	- May require removal of exotics and some replanting	- % natives per unit area - Species diversity
	Plant Size and/or Growth	- Tree girth (average and distribution of sizes)	- Annually pre and post restoration	- Vital habitat	- More closely resembles pre-disturbance or reference reach after restoration	- May require removal of exotics or watering and fertilizing early in the growth stages	- Annual increase in diameter
	Canopy Cover	- % of stream bed shaded	- Annually, during summer	- Food web support	- More closely resembles pre-disturbance or reference reach after restoration	- May need to plant additional woody vegetation or remove some trees if stream signficantly more shaded than reference condition	- % per area - Units of solar radiance
Physico-Chemical	**Surface Water Chemistry**	- N, P	- Regular intervals throughout the year (e.g., monthly, seasonally)	- Nutrient cycling - Food web support	- Conditions approach reference conditions - Meets or exceeds tolerance limits	- May need new or additional BMPs to minimize nutrient flux to the stream	- See Hauer and Lamberti, 2007 - Other methods available from EPA
	Temperature	- Temperature	- Regular intervals throughout the year (e.g., monthly, seasonally)	- Nutrient cycling - Food web support	- Conditions approach reference conditions	- May need canopy cover for shade (plant trees) or increased infiltration into soils and decreased overland flow to stream (especially if there is impervious cover nearby)	- Maximum and minimum daily temperatures generally more important than average temperature
	Oxygen	- Concentrations (mg/L)	- Regular intervals, particularly at night when photosynthesis not occurring	- Vital habitat	- Conditions should approach reference conditions	- Dissolved O_2 is not a common problem in streams unless there are large pools or prolonged low discharge (e.g., due to surface water extractions)	- Using continuous O_2 concentration measurements calculate diurnal variation in O_2 (net daily metabolism is the net O_2 change per day)
	pH	- pH	- Measure at regular intervals pre and post restoration if stream is impaired due to acid drainage or other source of low pH	- Vital habitat	- Conditions should approach reference conditions	- If acid-drainage source cannot be removed, liming of the stream may be possible or constructed wetlands may resolve problem (see	- Minimum and mean pH

Table 4. Stream Restoration Metrics Table.			
Category	**Metric**	**Basic measurement protocols**	**Additional Information and/or advanced measurement protocols**
Hydrologic	**Discharge**	-If a USGS gage station is located on the same stream and reasonably close to the project site, public data available from the National Water Information System Web Interface (NWISWeb) may be useful. Timeseries data can be downloaded at: http://waterdata.usgs.gov/nwis/ -If there is no USGS gage station nearby, a hydrograph can be created with time series data from a pressure transducer and a rating curve -For a basic explanation of rating curves, see: "Determining a Continuous Record of Discharge" in Wahl et al., 1995 http://pubs.usgs.gov/circ/circ1123/collection.html#HDR11 -Note that a useful rating curve requires numerous discharge measurements over a broad range of flow, preferably including low and high (i.e., storm) flow data	-For comprehensive information on the use of pressure transducers, see: Freeman et al., 2004 http://pubs.usgs.gov/twri/twri8a3/pdf/twri8-a3.pdf -For comprehensive information about stage and discharge measurement, see: Rantz et al., 1983 http://pubs.usgs.gov/wsp/wsp2175/
Geomorphic	**Substrate Particle Size**	-Protocol depends on stream's typical particle size (i.e., sand or gravel). -For projects with substrate coarser than sand, use the "Wolman pebble count" method, as in: Harrelson et al., 1994, p 49 to 51 http://www.stream.fs.fed.us/publications/PDFs/RM245E.PDF -Or: U.S. EPA, 2004, p 82 to 86 http://www.epa.gov/owow/monitoring/wsa/wsa_fulldocument.pdf -For finer particle sizes, use sieving, as in: Fischenich and Little, 2007, p 7 http://el.erdc.usace.army.mil/elpubs/pdf/sr39.pdf	-For more discussion of particle size measurement, see: Gordon et al., 2004, p 115 to 119
	% Fines	-For a basic visual estimate method, see: Sylte and Fischenich, 2002, Table 3 http://el.erdc.usace.army.mil/elpubs/pdf/sr36.pdf	-For a more quantitative method, grain size should be determined by submitting a sample to a laboratory for analysis (e.g., hydrometer method)
	Bed Armouring	-Protocol as in "substrate particle size " above then calculate the ratio of surface to subsurface median size	-More detailed information on streambed armoring and particularly the problem of determining dynamics during flood events are available in the literature (e.g., Wilcock and DeTemple, 2005)
	Floodplain Accretion	-Use feldspar markers, as described in Noe and Hupp, 2009, p 729 to 730 http://water.usgs.gov/nrp/proj.bib/Publications/2009/noe_hupp_2009.pdf	

Table 4. Stream Restoration Metrics Table.

Category	Metric	Basic measurement protocols	Additional Information and/or advanced measurement protocols
	Floodplain connectivity	-Crest-stage recorders - see USG methods (http://pubs.usgs.gov/twri/twri3a7/pdf/twri_3-A7_d.pdf) and other sources	- See notes above on flow regime and discharge; methods are basically the same except instead of calculating discharge, focus is on duration and frequency of overbank flows
	Channel Geometry	-See methods in: Harrelson et al., 1994. http://www.stream.fs.fed.us/publications/PDFs/RM245E.PDF	-This may only be useful for certaing goals.
	Large/Coarse Woody Debris	- Simple survey methods with counting per unit of stream or photographs are adequate	
Biota	**Species Lists (macroinvertebrates, fish)**	-See procedures in Hauer and Resh, 2006, p 435 to 437; or -See procedures and discussion of metrics in Barbour et al., 1999, chapter 7 http://water.epa.gov/scitech/monitoring/rsl/bioassessment/ch07main.cfm -For tolerance values, see Carter et al., 2007, p 833	- If data on the regional species pool are available (i.e., taxa expected at least impacted reference sites) then an advanced method is preferred that measures taxonomic completeness (observed/expected species), see Hawkins et al., 2000 and later papers - This can be relatively costly depending on available personnel. If costs prohibit a full taxa list, focus on sensitive species of fish and macroinvertebrates only; for example: Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) that are found in least disturbed reference sites
	% Sensitive Taxa (macroinvertebrates, fish)	-See procedures in Hauer and Resh, 2006, p 435 to 437; or -See procedures and discussion of metrics in Barbour et al, 1999, chapter 7 http://water.epa.gov/scitech/monitoring/rsl/bioassessment/ch07main.cfm -For tolerance values, see Carter et al., 2007, p 833	- New methods are emerging on the sensitivity of individual species to various stressors (e.g., Cuffney et al., 2010; King et al., 2011)
	Fish Size Structure	- Fish collected using nondestructive methods (e.g., electroshocking), identified, then measured for length and released	

Table 4. Stream Restoration Metrics Table.

Category	Metric	Basic measurement protocols	Additional Information and/or advanced measurement protocols
Riparian vegetation	**% Intact**		
	Width		- Advanced methods and information: Rheinhardt et al., 2009
	Composition	- Most states have lists of native/invasive species	- Advanced methods and information on Cheapeake Bay riparian buffer effectiveness: Orzetti et al., 2010
	Plant Size and/or Growth	- Establish transects, mark trees, and revisit them to measure change in girth using tape measure	- Platts et al., 1987 http://www.fort.usgs.gov/Products/ProdPointer.asp?AltID=1052
	Canopy Cover	- Measure canopy shading visually, with lens, or spherical densiometer	- Davies-Colley and Payne, 1998 http://www.jstor.org/stable/1467966
Physico-Chemical	**Surface Water Chemistry**	-In situ measurements with electronic meters (such as those manufactured by Hydrolab or YSI) are easiest -See details in FISRWG, 1998, p 7-68 to 7-69 http://www.nrcs.usda.gov/technical/stream_restoration/newtofc.html	- Advance methods include development of a nutrient budget or direct measurements of nitrogen or phosphorus uptake using tracer injections; see Hauer and Lamberti, 2007, various methods chapters
	Temperature	- Thermometer, thermistor array, or Mini-sonde with temperature probe	Hauer and Hill, 2007
	Oxygen	- Dissolved oxygen probe such as YSI or for continuous recording use various Mini-Sondes available for purchase that measure oxygen and other parameters such as temperatue and pH	- Whole stream metabolism (mg/O_2/day) is an advanced method in which oxygen concentrations are measured continuously over a 2 week period (using a deployed Mini-Sonde); levels should be within the range of reference sites. See Bott, 2007
	pH	- Negative H ion concentration using a pH meter	- Many methods are currently used to treat acidic streams; see article by Ormerod and Durance, 2009 for a discussion and examples

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Prioritized Sets of Recommended Metrics

Resources will limit total monitoring effort at a site and goals will suggest which metrics are most useful to measure. The list below represents a prioritization of metrics to be used in stream restoration. To promote data collection that allows restoration to be tracked and compared at multiple sites, we recommend a "Primary" set of metrics that addresses the most fundamental stream functions and processes. The "Secondary" set represents additional metrics that, depending on the restoration goal, would support multiple processes and functions. The "secondary" set of metrics provides options for measuring goal-driven outcomes, such as restoration of a particular species or ecosystems. Table 5 provides further description of the metrics, including recommended measurement protocols, but some aspects of measurement and interpretation will need to be tailored to project design and goals. Many metrics are only useful if they can be sampled at a sufficient number of locations throughout the restoration site and at sufficient frequency. Therefore, intensively sampling a few metrics, rather than superficially sampling many metrics, is likely to be a preferred strategy for monitoring effectiveness, particularly for larger sites.

Table 5. Recommended Monitoring Metrics

Constraint Metrics[1]
1. Land cover
(% deforestation/imperviousness/agricultural land use in watershed draining to the project site)
2. Upstream water quality (oxygen, suspended sediment, temperature)
3. Number of Dispersal barriers (recolonization potential)
Primary Metrics – Required by all projects and represent core structure & functions
1. Native riparian vegetation (% intact, composition & size)
2. Discharge
3. % fines on streambed
4. Bed Armoring
5. Water chemistry (select parameters based on putative stressor)
Secondary Metrics – Dependent on project goals
1. Substrate particle size
2. Species list or sensitive taxa list
3. Channel morphology
4. Large/coarse woody debris
5. Fish size distribution (if project is to restore fishery)
[1]Constraint metrics should be evaluated prior to the restoration and then post-restoration if there are major changes in the watershed likely to affect the project outcome OR if the project itself was specifically addressing the constraint (e.g., reforestation is a useful way to restore a stream over the long term)

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Appendix A. Evidence-Based Review of Performance Measures for Assessing Stream Restoration Projects

In this chapter, we examine the scientific evidence supporting the use of various metrics for evaluating the functional performance of stream restoration projects. Stream ecosystems generate a variety of processes that promote the restoration goals of reducing the flux of sediments and nutrients toward the Chesapeake Bay and providing vital habitat. These processes include: nutrient and sediment trapping or complete removal via denitrification, nutrient uptake by riparian plants, and floodplain accretion that stores sediment and phosphorus. Flow regime, in combination with reference-site 'levels' of sediment and larger substrate inputs, support the development and maintenance of habitat. The relationships between performance metrics (listed in Table 4-1) and the ability of a project site to generate these and other functions are discussed below; the strength of the evidence is summarized in Table 4-2. We indicate which metrics offer the strongest support for specific goals since metric selection and interpretation is guided by restoration project goals.

Table 4-1. Metrics reviewed for monitoring success of stream restoration projects.

Hydrology and Geomorphology	Biota	Physico-Chemical
• Discharge regime (magnitude, timing, and duration of high and low flows) • Channel geometry • Substrate characteristics (particle size, % fines, etc.) • Floodplain accretion • Suspended sediment dynamics • Presence of large woody debris	• Number of taxa • Presence or abundance of sensitive or tolerant taxa or species of interest • Biomass • Chlorophyll *a* • Intact native riparian vegetation	• Organic matter availability • Canopy cover (light levels) • Stream metabolism • Nutrient dynamics – pH, temperature, etc. • Habitat

Hydrology, geomorphology, and habitat

Discharge and sediment dynamics influence virtually every aspect of stream ecological condition. Sediment inputs, streambank and streambed erosion and deposition, and discharge magnitude (particularly the timing and duration of extreme events) control biogeochemical processes and the survival, growth, and reproduction of biota and plant communities. Thus, hydrogeomorphic processes can profoundly influence water quality and habitat. Since these two features are the primary focus of this report (i.e., NFWF's 'endpoints'), we focus mostly on their link to hydrogeomorphic processes and not on metrics that evaluate restoration of geomorphic structure. We do, however, acknowledge that there is a growing literature on restoration design and monitoring from a primarily hydrogeomorphic perspective (e.g., Doyle et al. 2007, FISRWG 2001). Dynamic measures such as bed stability, sediment flux, changes in grain size or sorting, and changes in riparian or in-stream vegetation provide useful insights into the geomorphic restoration trajectory if they are placed in the broader watershed and geologic context (Brierley and Fryirs 2005).

Discharge regime

It is important to recognize that the flow *regime* – magnitude, timing, and duration of discharges –exerts control over ecological recovery following restoration. Measuring discharge, either on a single or a few dates, provides insufficient information on stream ecosystems and their response to restoration. Fortunately, many inexpensive and easy to deploy automated tools are available to measure discharge over time (Gore 2006; pages 127-128 *in* Dahm et al. 2006). For example, pressure transducers with data loggers allow for the collection of long-term stage level data in an automated fashion. Ideally, stage data will be used to develop rating curves for the site so that actual discharge values (e.g., in ft^3/sec) can be estimated. However, even when rating curves are not developed for a particular site, relative changes in discharge (or stage) pre- vs. post-restoration or in restored vs. control (unrestored) or reference sites (least-disturbed) can be very informative. For example, Filoso and Palmer (2011) found that annual hydrographs for restored urban streams that retained more nutrients and sediments post-restoration showed attenuation of peak discharges post- compared to pre-restoration.

Neither point-in-time discharge measurements nor average annual discharge are good predictors of ecological patterns and processes (Bragg et al. 2005). **It is the hydrologic *regime* (the timing, magnitude, and duration of discharges) that shape stream ecosystem dynamics and biodiversity** (Poff et al. 1997, Postel and Richter 2003, Lytle and Poff 2004)**.** Restoration efforts should always seek to re-establish the historic hydrologic regime to the greatest extent possible. **In general, the seasonality, magnitude, and duration of peak and low flows, along with water quality, are the most important factors shaping the ecology and thus restoration potential of streams (Arthington et al. 2010)**. While these flow attributes are linked directly to biota and ecosystem processes (e.g., survival and reproduction), flow regime exerts significant indirect effects because of its influence on geomorphology, sediment transport, food availability, and habitat.

Urban and agricultural land use conversion have severe consequences for stream and watershed hydrology (Allan and Castillo 2007, Walsh et al. 2005), and several studies of stream restoration within human-impacted watershed have noted that an altered flow regime played a role in limiting restoration project success (e.g., Kasahara and Hill 2006, Quinn et al. 2009). Understanding stream processes at a range of discharges is important as a majority of both sediment (Allan and Castillo 2007) and nitrogen (Buffam et al. 2001) may be transported during high flows.

Water Quality and Vital Habitat. Evidence for the ecological influence of discharge dynamics is extensive (for a recent review see Poff et al. 2010). Hydrologic regimes influence macroinvertebrate (Boulton et al. 1992, Valentin et al. 1995) and fish (Gehrke et al. 1995, Kennard et al. 2007, Lammert and Allan 1999) community structure, channel geomorphology and substrate composition (Allan and Castillo

2007, Hammer 1972, Jowett and Biggs 1997, Morisawa and LaFlure 1979), the availability of basal resources (i.e., algae and organic matter; Bis et al. 2000, Ehrman and Lamberti 1992, Grimm and Fisher 1989, Jones and Smock 1991, Jowett and Biggs 1997, Lepori et al. 2005, Valetin et al. 1995), and sediment and nutrient delivery (e.g., Chien 1985, Jordan et al. 1997), among others. Rates of whole stream metabolism also vary with discharge (Roberts et al. 2007, Ryder and Miller 2005), as do nutrient uptake metrics (Mulholland et al. 2001). Hydrology also plays a major role in connecting stream channels to their adjacent floodplains. Hydrologic connections may influence riparian vegetation (Shafroth et al. 2002), use of floodplain habitat by fish (Junk et al. 1989), and nutrient removal processes (Craig et al. 2008).

A number of different metrics have been developed to evaluate the extent to which the hydrologic regime for a stream diverges from the historic (unimpacted) regime. For example, Richter et al. (1997, 1998) developed the “Range of Variability Analysis” that uses flow data series from a reference (unimpacted) stream to derive a target range for each of 32 flow metrics. In arid regions or areas where extractive losses of water are high (e.g., agricultural areas), stream restoration has focused as much on water quantity (minimum flows needed ecologically) as discharge variability, simply because the streams are essentially water-starved. For example, work in Australia estimated that Queensland Rivers needed approximately 80–92% of natural mean annual flow to maintain a low risk of environmental degradation (Arthington and Pusey 2003).

In addition to enhancing fish passage, restoration of flow regimes is also a major motivation for many dam removal projects. Dams not only fragment rivers longitudinally; they also reduce flow amplitude and the downstream flux of sediments, increase baseflow variation, and moderate temperature, all of which can have dramatic impacts on biota and ecological processes (Hart et al. 2002). If dams cannot be removed, the best option is to alter flow regimes by managing reservoir releases so that peak flows (at the right time of the year) are restored and baseflows are stabilized (Stanford et al. 1996). Today, there are a number of sophisticated methods for determining the range of flows and flow variability to target for restoration (Richter et al. 1997, 1998).

Substrate Characteristics (Bed sediment size, % fine-grained particles)

Vital Habitat – Substrate characteristics have been shown to influence biota, including algae (Hill et al. 2003), macroinvertebrates (Angradi 1999, Barlaup et al. 2008, Collier 1995, Duan et al. 2009, Dunbar et al. 2010, Lammert and Allan 1999, Longing et al. 2010, Maul et al. 2004, Miserendino 2001, Monaghan and Soares 2010, Richards et al. 1993, Rempel et al. 2000, Roy et al. 2003), and fish (Berkman and Rabeni 1987, Frimpong et al. 2005, Pires et al. 2010, VanDusen et al. 2005, Walters et al. 2003, Wood and Armitage 1999). Typically, increased quantities of fine sediments, such as those found in human-impacted settings, negatively impact biotic communities, but results often depend on species or functional group. Substrate heterogeneity (e.g., variability in particle size of the sediments, D_{84}/D_{50}) has been used as a metric to evaluate the quality of habitat in streams under the assumption that higher substrate heterogeneity should lead to higher biological diversity (for a review of this topic, see Palmer et al. 2010). However, to date, manipulations of in-stream substrate heterogeneity have not routinely resulted in a positive biotic response (Brooks et al. 2002, Pretty et al. 2003, Lepori et al. 2005)

Streambed armoring can influence habitat quality for benthic species and fish who deposit eggs below the streambed surface because the exchange rate of water and oxygen and the movement of subsurface sediment may be impeded by the larger substrate particles on the streambed surface (Allan and Castillo 2007). A measure of the coarseness of the bed surface particles compared to the material underneath (the ratio of the median particle size, D50 surface to D50 subsurface) can be used to quantify armoring (Bunte and Abt 2001). The amount of readily available particles that would otherwise be transported is reduced by armoring, and thus the streambed traps sediments, often becoming embedded in a "pavement like state" (Lisle and Church 2002, Encinas and Meyers 2010).

Water Quality – Sediments dominated by silts and clays may influence dissolved nutrient concentrations, especially phosphorus which sorbs to fine sediments. For this reason, abiotic uptake of P may increase in streams with finer substrate (Meyer 1979, Ryan et al. 2007). For example, Meyer (1979) noted that P sorption increased with decreasing sediment particle size in a study conducted in the Hubbard Brook Experimental Forest, New Hampshire. Moreover, an excess of fine sediments may prevent exchange at the streamwater-groundwater interface (i.e., the hyporheic zone), which can result in decreased retention and transformation of nutrients. For example, Kasahara and Hill (2006) linked areas of decreased hydraulic conductivity associated with presence of fine sediments to reduced N uptake in a study of constructed riffles in agricultural and urban streams of southern Ontario, Canada.

Channel Morphology (Cross-sectional profile, width, depth, sinuosity)

Water Quality and Vital Habitat – Measures of channel geometry (i.e., cross-sectional profile, width to depth ratio, sinuosity) are often used to assess changes in channel stability in response to land use change or a restoration intervention. Urban land use and the resultant increase in the amount of water delivered via overland flow paths generally causes channels to become wider and less sinuous (Hammer 1972, Morisawa and LaFlure 1979, Pizzuto et al. 2000, Walsh et al. 2005). The presence or absence of riparian vegetation may also influence channel geometry. For example, Sweeney et al. (2004) found that streams in Pennsylvania with forested riparian zones were wider and had increased bed roughness and lower average water velocity than deforested channels. Quinn et al. (1997) similarly observed that forest streams were wider than unshaded, pasture streams in New Zealand. Changes in stream morphology can be used to estimate both sediment transport and channel stability. For example, Trimble (1997) used changing measures of channel geometry in California to estimate that significant quantities of sediment (106,000 Mg yr^{-1}) were eroded from an urbanizing stream in southern California over a ten-year period. Smith and Prestegaard (2005) used extensive channel geometry measurements to identify channel instability as the cause of failure of a stream restoration project in Maryland.

Several studies have related measures of channel geometry to biotic responses. For example, macroinvertebrate community metrics, including diversity and composition, have been correlated to channel instability (Death and Winterbourn 1995), width to depth ratios (Hutchen et al. 2009), and cross-sectional diversity (Nakano and Nakamura 2008), though the direction of these relationships was often species- or context-dependent. Changes in channel geometry over time can be useful in evaluating coarse-scale geomorphic response to restoration, however alone they do not provide readily interpretable ecological information. When measures of structural attributes are directly tied to underlying hydrogeomorphic processes, they can be useful in combination with other metrics to determine restoration trajectory. For example, in-stream and riparian vegetation structure as well as woody debris all reflect system condition and are tied to or directly influence stream processes. In general, as emphasized previously, measures of the processes that shape stream ecosystem structure and function are more useful in evaluating restoration projects (Brierley et al. 2010).

Flood Plain Accretion

Water Quality – Stream and river floodplains are recognized sites of retention for sediment (Steiger et al. 2003) and sediment-bound phosphorus (Noe and Hupp 2005). Spatial patterns of sediment accretion are influenced by the presence of woody debris and floodplain vegetation, which can also impact local water velocities, however, floodplain sediment storage is controlled primarily by sediment supply and stream hydrology (Gurnell 1997, Jeffries et al. 2003). While floodplain attributes are commonly documented during physical habitat assessments to gauge the potential for floodplain storage, these surveys require assumptions about the previous state of the floodplain and do not provide a quantitative assessment of sediment accumulation (Steiger et al. 2003). Artificial turf mats or feldspar pads, however, do provide direct, quantitative assessments of sedimentation rates over time (Noe and Hupp 2009, Steiger et al. 2003).

Physical Habitat

Vital Habitat –Physical habitat assessment protocols vary widely across the United States, but generally include a suite of structural attributes believed, as an aggregate, to represent overall habitat quality (Whitacre et al. 2007). In this region, a physical habitat index created using data from the Maryland Biological Stream Survey included measures of bank stability, instream wood, epibenthic substrate, shading, remoteness, riparian width, and embeddedness, among others (Paul et al. 2002).

Several studies have observed correlations between habitat indices and bioindicator metrics (e.g., Monaghan and Soares 2010, Sullivan et al. 2004, Sullivan and Watzin 2008). For example, Sullivan and Watzin (2008) observed a significant positive relationship between benthic macroinvertebrate and fish metrics and physical habitat condition scores. However, significant correlations between biotic metrics and physical habitat conditions are typically confounded with water quality condition (i.e., streams with degraded habitats also typically have high sediment loads and/or pollutants).

In addition, a similar number of studies have shown a lack of correlation between biotic metrics and physical habitat assessment scores (e.g., Frimpong et al. 2005, Hannaford and Resh 1995, Monaghan and Soares 2010). For example, in an assessment of restored, unrestored, and reference streams in California, Hannaford and Resh (1995) reported a poor relationship between physical habitat assessment scores and macroinvertebrate metrics, though in a related study, Purcell et al. (2002) found there was agreement between the rank of physical habitat scores and EPT taxa scores. Frimpong et al.

(2005) similarly reported that fish IBI's were more poorly correlated to qualitative habitat index (QHEI) scores; they found that only two characteristics included in the index were significantly related to fish IBI: reach scale substrate and riffle-pool quality.

The lack of consistent evidence that higher habitat index scores are associated with biological improvements (e.g., increased diversity) calls the utility of the habitat metric into question – clearly factors other than habitat alone are critical (Tullos et al. 2009, Miller et al. 2010, Bernhardt and Palmer 2011). In particular, a recent study that evaluated 78 stream restoration projects with enhanced habitat structure (e.g., by increasing the number and diversity of habitats), found only two projects with statistically significant increases in macroinvertebrate diversity (Palmer et al. 2010). Work to date suggests that in order to restore stream biotic communities, the most limiting factors must be addressed first, and typically it is not degraded habitat. If the processes that *cause* habitat degradation are restored (e.g., riparian plant production, discharge regime) and water quality is not impaired for other reasons (e.g., pesticide or fertilizer run-off), then habitat and biota should respond positively. However, enhancing physical habitat only, for example by adding boulders or woody-debris, rarely results in the restoration of stream biota. After monitoring the response of 26 pairs of restored and non-restored sites in Europe, Jahnig et al. (2010) concluded that restoring habitat on scales much larger than single reaches, in combination with measures to tackle catchment-wide problems (e.g. water quality, source populations), is required for a recovery of stream communities.

Large Woody Debris (LWD)

Vital Habitat – Addition of wood to streams is a common practice used worldwide in stream restoration. Since wood, like any flow obstruction, plays a key role in shaping velocity and sedimentation profiles, forming pools, and shaping banks, and may increase habitat heterogeneity and faunal diversity, how and where wood is placed is given careful consideration in restoration projects (Miller et al. 2010). Wood also provides habitat for fauna, substrate for biofilms, and refuge from predators and flow extremes (Benke and Wallace 2003). LWD has also been positively correlated with organic matter retention (Ehrman and Lamberti 1992, Raikow et al. 1995). However, for a number of stream restoration projects involving LWD additions (reviewed by Entrekin et al. 2009), organic matter availability did not increase following restoration. While LWD has been positively correlated to macroinvertebrate taxa richness (Gerhard and Reich 2000, Johnson et al. 2003, Hrody et al. 2008, Lester and Boulton 2008) and density (Gerhard and Reich 2000, Moerke et al. 2004), some studies have found highly variable responses by invertebrates, including no increase in abundance (Larson et al. 2001), no change in secondary productivity (Entrekin et al. 2009), no net change in taxa composition (Wallace et al. 1995), or even a net decrease in some taxa (Hilderbrand et al. 1997). In a paper summarizing results of a mail survey related to 50 wood addition restoration projects in Europe , Kail et al. (2007) report that the effects of wood structures were strongly dependent on conditions such as stream size and hydrology, which probably explains the variability in response to wood additions.

Large woody debris has been positively correlated with fish and amphibian community metrics (e.g., Everett and Ruiz 1993, Richmond and Fausch 1995, Wilkins and Peterson 2000); however, some LWD addition studies found little or no effect on fish abundance (Sweka and Hartman 2006). In cases where fish abundance increased following additions of in-stream structures, it is not clear if structures merely aggregated (attracted) individuals or actually increased production (Roni et al. 2005, Palmer 2009). Because the effects of LWD additions during restoration projects are highly variable, Kail et al. (2007) and Shields et al. (2008) emphasized the importance of restoring the natural processes the lead to the presence of LWD in streams; they emphasized that highly-engineered designs that fix logs in place are expensive and have a failure rate that may require repeated inputs.

Water Quality – LWD has been positively correlated with hydraulic residence time (Ehrman and Lamberti 1992, Ensign and Doyle 2005, Roberts et al. 2007) and phosphorus (Ensign and Doyle 2005, Valett et al. 2002) and ammonium (Roberts et al. 2007) uptake. The impacts of LWD on sediment accumulation are less clear, with some evidence showing that LWD has no impact on sediment accumulation (e.g., Larson et al. 2001), that sediment may be only temporarily retained by LWD (e.g., Nakamura and Swanson 1993), or that sediment may be retained by fine woody debris (Bilby and Ward 1991).

Biota

Macroinvertebrates

Water quality and vital habitat – Benthic macroinvertebrate communities are believed to provide an aggregate view of in-stream conditions because they respond to a variety of different stressors and integrate the impacts of those stressors over time (Barbour et al. 1996, Bonada et al. 2006). However, communities can only serve as 'indicators' of the factors they are sensitive to, such as low oxygen levels for extended periods of time or significant siltation. They may not be reflective of some conditions that are critical to Chesapeake Bay health. For example, they are insensitive to elevated nitrogen loads unless the N concentration is extremely high for a sustained period of time (Yuan 2010). However, an abundant and diverse macroinvertebrate community appears important to healthy freshwater fisheries (Allan and Castillo 2007). A number of macroinvertebrate bioassessment tools have been developed and they typically employ a suite of metrics that may include measures of richness (e.g., number of total taxa, number of EPT taxa, i.e., Ephemeroptera, Plecoptera & Trichoptera), composition (e.g., % EPT taxa, % Diptera), and/or tolerance (e.g., % dominant taxon, % tolerant taxa; Barbour et al. 1999). Many studies have linked one or more of these metrics to land use, and more generally, stream condition (e.g., Barbour et al. 1996, Hawkins et al. 2000, Moore and Palmer 2005, Purcell et al. 2009, Stephenson and Morin 2009, Walsh et al. 2005, Walters et al. 2009).

Macroinvertebrate community metrics are correlated with water quality variables including temperature (Castella et al. 2001, Collier 1995, Faith and Norris 1989, Jacobsen et al. 1997, Miserendino 2001, Sponseller et al. 2001, Sweeney 1993, Wohl et al. 1995), conductivity (Collier 1995, Echols et al. 2009, Marchant et al. 1997, Maul et al. 2004, Miller et al. 2007, Miserendino 2001, Roy et al. 2003, Walters et al. 2009), metals (Clements 1994, Clements et al. 2000, Hirst et al. 2002, Mebane 1999, Rhea et al. 2006), and nutrient concentrations (Bernatowicz et al. 2009, Bis et al. 2000, Faith and Norris 1989, Heino et al. 2003, Hirst et al. 2002, Justus et al. 2010, Malmqvist and Maki 1994, Maul et al. 2004, Roy et al. 2003, Smith et al. 2007, Wang et al. 2007). However, as noted above, they cannot be used to diagnose a specific problem such as metal contaminants or high nutrient levels because these pollutants are almost always associated with multiple, confounding effects. Consider, for example, that streams with high nutrient concentrations, such as those in agricultural areas, typically also have high levels of siltation and pesticides, making it difficult to use macroinvertebrate assemblages to pinpoint specific stressors.

Sub-metrics of community indices also respond to variables that more broadly influence habitat type, including substrate type (Angradi 1999, Collier 1995, Duan et al. 2009, Dunbar et al. 2010, Faith and Norris 1989, Lammert and Allan 1999, Maul et al. 2004, Mebane 1999, Miserendino 2001, Monaghan and Soares 2010, Paul et al. 2002, Rempel et al. 2000, Richards et al. 1993, Roy et al. 2003, Walters et al. 2009) and benthic organic matter availability (Bis et al. 2000, Culp et al. 1983, Faith and Norris 1989, Rempel et al. 2000, Shieh et al. 1999). Some studies have found correlations between indices of habitat quality and macroinvertebrate metrics (e.g., Paul et al. 2002, Sullivan et al. 2004, Sullivan and Watzin 2008), while others have not (e.g., Hannaford and Resh 1995, Monaghan and Soares 2010, Purcell et al. 2002).

In general, studies have not been able to link index scores to specific stressors, making it difficult to interpret metrics because communities can show impairment due to a stress (e.g., hydrologic regime) that is different from the stress being evaluated (e.g., water quality). Therefore, once an impairment is detected using macroinvertebrate community metrics, additional data, including detailed measurements of physical and chemical characteristics, are required to identify the variables that are most influencing macroinvertebrate response and, ultimately, for implementing appropriate mitigation or adaptive restoration strategies (Barbour et al. 1996). Thus, as others have emphasized, use of macroinvertebrates as indicators for stream restoration is not recommended unless the stream was only very slightly impaired prior to restoration. Similar to other biotic indicators (e.g., fish, algae), macroinvertebrate data provide information on the *condition* of a stream but insufficient information on the underlying pressures that have led to a degraded state (Brierley et al. 2010).

Fish

Water quality and vital habitat – Fish bioassessments are frequently conducted because of the cultural and economic importance of these animals (Barbour et al. 1999). Like macroinvertebrate bioassessments, metrics based on fish community assemblages are often used to evaluate 'ecological integrity', however, responses to individual stressors have also been described (Karr and Chu 1999). Numerous studies have related fish metrics to land use (e.g., Frimpong et al. 2005, Kennard et al. 2007, Lammert and Allan 1999, Lenat and Crawford 1994); urban land use appears to have particularly detrimental impacts on fish assemblages due to alteration of hydrological regimes, temperature, and water chemistry (Lenat and Crawford 1994, Wang et al. 2001).

Fish community metrics have been correlated with several water quality variables including nutrients (Justus et al. 2010, Lenat and Crawford 1994, Wang et al. 2007), metals (Lenat and Crawford 1994, Mebane 1999), and total suspended sediment, which may lead to gill damage, decreased growth rates, and lower reproductive success (Burkhead and Jelks 2001, Lenat and Crawford 1994, Sutherland and Meyer 2007). Because the impacts of water quality on specific fish metrics (e.g., taxa richness or abundance) often vary by species or trophic level (e.g., herbivores, detritivores), relationships between water quality and multi-metric fish indices may be poor (Walters et al. 2009). Some specific fish guilds are particularly sensitive to specific stressors and may serve as indicators of that stress if they are the only guild responding (e.g., salmonids are especially sensitive to elevated temperatures).

Fish community metrics are also related to variables that broadly influence habitat quality including hydrological regime (Kennard et al. 2007, Lammert and Allan 1999), availability of food resources (e.g., algae or macroinvertebrates; Griffith et al. 2009, VanDusen et al. 2005), substrate type (Frimpong et al. 2005, Pires et al. 2010, Walters et al. 2009), and canopy cover (Jones et al. 1999, Jowett et al. 2009, Kennard et al. 2007, Penczak 1995, Pires et al. 2010). Again, effects are often species-specific. However, fish abundance (Berkman and Rabeni 1987, Mebane 1999), density (Berkman and Rabeni 1987, VanDusen et al. 2005), and biomass (VanDusen et al. 2005) tend to decrease with increasingly fine substrate. Fish abundance and richness similarly decrease with reductions in canopy cover (Jones et al. 1999, Jowett et al. 2009, Penczak 1995, Pires et al. 2010). Some studies report correlations between fish and habitat quality indices that incorporate many of these factors (e.g., Monaghan and Soares 2010), while others have not (e.g., Frimpong et al. 2005), however correlations between fish metrics and physico-chemical variables do not indicate causation. Therefore, interpretation of fish community metrics also requires additional information (e.g., chemical or physical metrics) so that factors underlying fish community response can be identified (Suter 1993). While less common in the Chesapeake Bay watershed compared to the Pacific Northwest, many stream restoration projects have been completed to enhance habitat for fish (Roni and Beechie 2008). However, with the exception of a few restoration studies on salmon, positive responses to in-stream habitat "improvements" are rare and it remains unclear if reach-scale increases in fish abundance are due to aggregation (e.g., attraction to structures) or overall increases in fish productivity (Whiteway et al. 2010).

Periphyton

Water Quality and Vital Habitat -- Algal-based metrics used to assess overall water quality and ecological condition rely, at least in part, on taxonomic identification (e.g., periphyton indices of biotic integrity, PIBI's). Periphyton assemblages are more sensitive to nutrient loading, metals, and herbicides than fish or macroinvertebrates (Resh 2008) and communities often indicate impacts that can only be indirectly observed using assessments that focus on higher trophic levels (Barbour et al. 1999). However, as was the case for invertebrates and fish, many factors influence algal abundance and diversity, making it difficult to identify the primary stressor (i.e., their use in identifying what a restoration project should seek to 'fix').

PIBI's have been correlated with water quality parameters including pH, conductivity, metals, nutrients, and other major dissolved ions (Griffith et al. 2009, Hill et al. 2000, 2003, Hirst et al. 2002, Justus et al. 2010, Zalack et al. 2010). For example, Justus et al. (2010) found that several periphyton metrics (e.g., abundance of tolerant diatoms, abundance of *Cymbella* spp., percent taxa richness) were negatively correlated with N and P concentrations.

Periphyton metrics are also related to variables that broadly influence habitat quality, including substrate size (Griffith et al. 2002, 2009, Hill et al. 2003), channel geometry (i.e., depth; Hill et al. 2000), and canopy cover (Bixby et al. 2009, Griffith et al. 2002, Hill et al. 2003). For example, Griffith et al. (2002) found that diatom species richness and species evenness were positively correlated with % slow water habitats and % sand and fines and negatively correlated to % cobble substrate and % canopy cover. Bixby et al. (2009) similarly found that both algal species richness and the densities of solitary diatoms (e.g. *Navicula* spp.) were lower in open canopy streams. As was the case for the macroinvertebrate and fish bioassessments, it is difficult to link periphyton metrics to factors that may be influencing the metric.

Algal Biomass (Chlorophyll *a*, Ash-free dry mass)

Water quality and Vital Habitat -- Algal biomass is measured as either chlorophyll *a* or ash-free dry mass (AFDM) and is influenced by a suite of factors including light levels, nutrients, flow regime, and temperature (Allan and Castillo 2007). Algal biomass has been shown to be negatively correlated with turbidity (Jacobson et al. 2008) and positively correlated with both iron concentration and stream water color (Hill et al. 2000). While some studies have shown a positive relationship between nutrient concentrations (i.e., N or P) and chlorophyll *a* concentrations (e.g., Rosemond et al. 1993), others have not observed a relationship (e.g., Bott et al. 2006, Jacobson et al. 2008).

Hydrologic regime and substrate stability also influence algal biomass, with lower chlorophyll *a* concentrations observed following high discharge events (Bis et al. 2000, Grimm and Fisher 1989, Jacobson et al. 2008). Algal biomass is also negatively impacted by the presence of grazing organisms (e.g., macroinvertebrate scrapers; Bis et al. 2000, Feminella et al. 1989, Ludlam and Magoulick 2009, Rosemond et al. 1993). Alternatively, biomass increases with decreased canopy cover (e.g., DeNicola et al. 1992, Jacobson et al. 2008), though the initial response to riparian restoration may be weak as adequate canopy cover takes time to establish (Quinn et al. 2009). Many of these factors are associated with land use, which may indirectly impact algal biomass (e.g., Stephenson and Morin 2009, Von Schiller et al. 2008).

The availability of algaealso influences upper trophic levels. For example, in a study conducted in the Little Miami River (Ohio) watershed, periphyton biomass was positively correlated to increased abundance of macroinvertebrates and fish; % herbivorous fish increased with increasing algal availability (Griffith et al. 2009). However, large quantities of algal biomass are undesirable, regardless of the positive influence of algae on stream biota (Welch et al. 1988). Studies on the response of algae to stream restoration are only now beginning (e.g., Meisenbach et al. 2010 ESA meeting abstract). Chlorophyll *a* concentrations are sometimes used as an indicator for gross primary production (GPP), though correlations between these metrics are typically poor (Izagirre et al. 2008, Ryder and Miller 2005, Young and Collier 2009). Total chlorophyll *a* per unit area is an example of a structural metric that has been used as an indicator of ecosystem function, though there is little evidence to support its use; for this reason, direct measures of ecosystem metabolism are more reliable (Ryder and Miller 2005).

Riparian Vegetation

We have identified two aspects of the riparian vegetative cover that are considered important in restoring a stream. The percent of the stream corridor from the bottom of the restoration reach to the uppermost part of the headwaters that is intact (**% Intact**) is probably the most important factor influencing water quality (assuming there are not point source inputs), habitat, stream temperature, and many other factors discussed below. In a study of 30 Chesapeake Bay streams with riparian vegetation ranging from 0 to 50 years of age, Orzetti et al. (2010) found that habitat, water quality, and benthic macroinvertebrate metrics improved with age of restored buffer. The importance of riparian vegetation to stream health is emphasized by the ample data on the composition of riparian vegetation within the Chesapeake Bay watershed (Rheinhardt et al. 1999; 2009). If an individual project only has the ability to restore the vegetation along a single reach and there are gaps in the vegetated corridor above that reach, then the project should report the % intact along the reach <u>and</u> the % intact above the reach; the second metric is a major constraining factor and may render the proposed project reach unsuitable for restoration investments. As discussed below, the composition of the vegetation (**% native vegetation**) and its size (**% canopy cover)** compared to reference reaches is also important.

Vital Habitat – Reductions in % canopy cover may lead to increased light penetration and subsequent increases in stream temperature (Gravelle and Link 2007, Sweeney 1993), altered availability of basal resources (i.e., algae vs. leaf litter; Bis et al. 2000, Gillig et al. 2009, Jacobson et al. 2008), and resultant shifts in food web structure. Specifically, decreased canopy cover has been correlated with increased algal biomass (as both chlorophyll *a* and AFDM), decreased algal species richness, and shifts in algal community structure (Bixby et al. 2009, DeNicola et al. 1992, Griffith et al. 2009, Hill et al. 2003, Jacobson et al. 2008, Sabater et al. 2000). Decreased canopy cover has also been correlated with reductions in allochthonous inputs and benthic organic matter availability (Reid et al. 2008), though even sites with similar canopy cover may vary with respect to the quantity of leaf litter inputs (Muenz et al. 2006).

Invertebrate assemblages may be indirectly affected by canopy cover through periphyton-mediated effects(Bis et al. 2000, Brainwood and Burgin 2006, Ludlam and Magoulick 2009, Progar and Moldenke 2009). For example, Bis et al. (2000) noted that the abundance of scrapers was significantly correlated with open canopy. Invertebrates are also affected by temperature-mediated effects. Specifically, stream warming may influence growth rates, survivorship, fecundity, and timing of emergence or reproduction (Albarino et al. 2008,Sweeney 1993). Fish communities are also impacted by canopy cover and low canopy cover has been correlated to reduced species richness, abundance, and biomass (Jones et al. 1999, Jowett et al. 2009, Penczak 1995, Pires et al. 2010).

Canopy cover has also been related to whole-stream metabolism. Streams with greater canopy cover, and therefore less light reaching the channel, are typically more heterotrophic than streams with little or no shading (Hill et al. 2002, Sweeney et al. 2004). Likewise, Bunn et al. (1999) found that % canopy cover was negatively correlated with both gross primary production (GPP) and P:R (photosynthesis : respiration) ratios.

The fact that canopy cover influences, or can be highly correlated with, basal food availability (algae and/or allochthonous inputs), water temperature, large woody debris, and biotic composition makes it a useful restoration metric. The goal should be to establish canopy cover similar to that in least disturbed reference sites, however, even if this is accomplished there may be many other important factors that limit restoration success.

Water Quality – VanDusen et al. (2005) observed that % fine sediment increased with decreased canopy cover, though this relationship reflected a lack of riparian vegetation to trap sediment before it entered the stream, rather than a direct effect. Sabater et al. (2000) observed that phosphorus uptake was greater in streams with lower % canopy cover as a result of greater algal demand (measured as algal biomass and % streambed covered with algae). Research has also shown that when canopy cover is decreased (e.g., at newly restored sites in which riparian trees were removed to get access to the stream), nitrogen concentration in the stream may decline due to an increase in algal growth or an increase in water temperature and light (Sudduth et al. 2011). However, this may also lead to algal blooms and subsequent die-offs that leave the water low in oxygen and/or result in a net export of nutrients (tied up in the algal biomass) during flood periods.

It is important to recognize that that groundwater flow paths may circumvent the riparian root zone (e.g., naturally or because the stream is incised), and when this happens high nutrient inputs from groundwater may lead to elevated stream nutrient concentrations despite an intact riparian corridor (e.g., Speiran 2010). Gaps in the riparian coverage above a restoration site can also to elevated nutrient levels in a restored reach that has a fully intact riparian corridor (Sutton et al. 2010). In general, understanding flow paths to streams is critical to identifying priority riparian restoration sites and in understanding why, in many cases, nutrient levels in streams may be high despite vegetative cover (Weller et al. *in press).*

Composition of the riparian vegetation can also influence ecosystem processes, such as decomposition, that in turn influence water quality (e.g., through an impact on organic matter form and availability; Kominoski et al. 2011)

Physicochemical Characteristics including ecological functions

Temperature, pH, conductivity, DO, TSS, nutrients

Vital Habitat – A variety of physicochemical parameters, including temperature, pH, conductivity, and dissolved oxygen are common monitoring metrics. Other important measures of water quality are recorded less often and include: total suspended solids (TSS), nutrient concentrations (e.g., N, P), metals (e.g. copper, lead), and other major ions (e.g., chloride, sulfate, etc.). These parameters are often correlated with biotic community composition and rates of leaf liter decomposition and metabolism. The predictive power of point-in-time physicochemical measurements is generally weak because of interactivities between several variables (e.g., canopy cover, presence of grazers, hydrologic regime) and the fact that biotic responses may be species or group specific. Regardless, physical and chemical attributes of streams provide contextual information that is useful for interpreting other monitoring metrics, may be helpful for identifying key variables that influence biotic responses, and ultimately, for implementing appropriate mitigation or adaptive restoration strategies (Barbour et al. 1996).

Water Quality – Several studies have observed a correlation between nutrient concentrations and nutrient uptake (e.g., Dodds et al. 2002, Earl et al. 2006, Webster et al. 2003). However, nutrient removal is influenced by a variety of other factors (e.g., metabolic state, channel geometry, organic matter availability, etc.), and therefore cannot be estimated from nutrient concentrations alone. Furthermore, while nutrient and total suspended sediment concentrations may serve as useful indicators of current water quality, these measurements vary greatly with hydrology and seasonality, making instantaneous measures of water quality of little use for monitoring restoration success. Repeated measures of water quality attributes collected as a part of an on-going post-implementation monitoring program that spans the full range of discharge and seasons, may provide more valuable information, particularly if measurements are made above and below restored reaches.

Benthic organic matter

Vital Habitat – Benthic organic matter (BOM) standing stocks (e.g., mg carbon/unit area or volume of sediment) are largely influenced by factors that affect organic matter inputs and retention (Jones 1997), both of which are negatively impacted by human-dominated land uses (Golladay et al. 1989, Meyer et al. 2005). BOM availability has been positively correlated to macroinvertebrate survivorship, growth, abundance, and diversity (Eggert and Wallace 2003, Negishi and Richardson 2003, Pretty and Dobson 2004). Wallace et al. (1999) also observed a positive relationship between BOM availability and secondary production in an Appalachian stream. Relationships between BOM and whole stream metabolism are less frequently considered, and results are inconclusive. Ecosystem metabolism has been shown to increase with increasing BOM in some studies (e.g., Acuna et al. 2004) but not others (Houser et al. 2005).

Water Quality – Numerous studies have looked at the impacts of BOM availability on nutrient removal. Uptake rates of ammonium (Meyer et al. 2005) and phosphorus (Aldridge et al. 2009, Meyer et al. 2005) have been shown to increase with increased BOM availability in some urban streams. Nitrate uptake is particularly high on organic substrates (BOM and coarse wood), which essentially act as hotspots for microbial activity (Hoellein et al. 2009). Denitrification has also been positively correlated with BOM availability (Arango et al. 2007, Kemp and Dodds 2002, Opdyke et al. 2006, Opdyke and David 2007).

Measures of BOM standing stocks have already been used to assess a number of restoration projects (e.g., Laasonen et al. 1998, Muotka and Laasonen 2002, Negishi and Richardson 2003, Suren and McMurtrie 2005, Tullos et al. 2009). For example, Negishi and Richardson (2003) observed significant increases in BOM one year after implementation of a geomorphic restoration project in British Columbia, Canada that aimed to increase in-stream heterogeneity. Similarly, Sudduth et al. (2011) found significantly higher levels of buried fine particulate organic matter in four restored urban streams compared to four degraded or forested streams. Alternatively, Tullos et al. (2009) observed no statistically significant differences in BOM availability between restored and unrestored reaches in the

North Carolina Piedmont. Because BOM influences stream nutrient dynamics and metabolism, use of BOM as a metric may be valuable in restored streams that have low levels of BOM prior to project implementation (e.g., streams subjected to high, scouring flows pre-restoration). However, organic matter levels should be restored to levels similar to those found in reference reaches. Levels of BOM higher than fully forested sites, such as that found by Sudduth et al. (2011), does not mean the restored stream is healthier. In fact, high levels of BOM could have negative consequences, if for example, it leads to high microbial respiration and depletion of oxygen.

Leaf-Litter and Decomposition

Vital Habitat – Leaf litter decomposition rates decrease in response to a range of stressors including increased concentrations of arsenic (Chaffin et al. 2005, Lecerf and Chauvet 2008), copper (Schultheis et al. 1997), zinc (Niyogi et al. 2001), pesticides (Schafer et al. 2007), and insecticides (Wallace et al. 1996, Whiles et al. 1993), as well as increased sedimentation (Sponseller et al. 2001) and decreases in pH (Dangles et al. 2004, Simon et al. 2009, Riipinen et a. 2009). Alternatively, decomposition rates often increase in high-velocity environments as a result of greater mechanical fragmentation (Chadwick et al. 2006, Imberger et al. 2008, Lepori et al. 2005, Paul et al. 2006). Decomposition rates also increase with increased in-stream nutrient concentrations in some studies (e.g., Grattan and Suberkropp 2001, Gulis and Suberkropp 2003, Gulis et al. 2006, Meyer and Johnson 1983, Robinson and Gessner 2000,Suberkropp and Chauvet 1995) but not others (e.g., Lecerf and Chauvet 2008, Pascoal et al. 2005, Royer and Minshall 2001).

The response of decomposition rates to current or former land-use is less clear. Rates have responded to increasing levels of urbanization (Chadwick et al. 2006, Imberger et al. 2008, Young and Collier 2009), though patterns are inconsistent across studies (Walsh et al. 2005) with some finding no relationship (e.g., Royer and Minshall 2001, Danger and Robson 2004, Hagen et al. 2006, Paul et al. 2006). Several authors attributed altered rates of processing to changes in the macroinvertebrate or microbial community (i.e., density, abundance, loss of key species, increased activity; Chaffin et al. 2005, Dangles et al. 2004, Paul et al 2006, Schafer et al. 2007, Sponseller et al. 2001), yet many recognized that direct measurements of leaf-litter decomposition rates were often more sensitive to impacts than macroinvertebrate and fungal community metrics (e.g., McKie and Malmqvist 2009, Riipinen et al. 2009, Simon et al. 2009). However, because leaf litter decomposition rates varied greatly by leaf species (Gessner and Chauvet 2002), species used for decomposition assays require careful selection.

Thus far, the few studies that used leaf-litter breakdown to assess restoration success found that some restoration approaches have little (e.g., boulder addition to channelized reaches, Lepori et al. 2005) to no (e.g., LWD additions, Entrekin et al. 2009) influence on decomposition rates. In general, the use of litter decomposition as a measure of stream health is difficult because many factors can influence the rate of litter breakdown. For example, excessive flows may increase the rate of breakdown compared to reference streams (indicating a need for restoration actions that reduce peak flows), but an increase in the number of macroinvertebrate 'shredder' species may also increase litter decomposition (which may be viewed as a positive outcome).

Stream Metabolism

Vital Habitat – Stream metabolism (gross primary production (GPP) and ecosystem respiration (ER)) measure the rates of carbon production and use within a stream ecosystem. Furthermore, because GPP, ER and GPP/ER respond to a number of variables, including light (Bott et al. 2006, McTammany et al. 2007, Roberts et al. 2007), temperature (Mulholland et al. 2001, Webster et al. 1995), discharge (Atkinson et al. 2008, Roberts et al. 2007, Uehlinger 2006), nutrient and organic matter availability (Acuna et al. 2004, 2007, Atkinson et al. 2008, Izagirre et al. 2008), and land use (Young et al. 2008), they provide an integrative functional measurement and an overall assessment of stream health (Bunn et al. 1999). However, the interpretation of a 'healthy' level of metabolism requires knowledge of variability in whole stream metabolism in reference streams (i.e., higher or lower metabolism is not necessarily a positive outcome). The goal should be to restore metabolism to levels comparable to healthy streams.

Water Quality – In addition to providing information on stream energetics, metabolism has also been related to nutrient dynamics in several studies (e.g., Fellows et al. 2006, Hall and Tank 2003, Hoellein et al. 2007, Webster et al. 2003). Measures of in-stream metabolism have only rarely been employed as part of stream restoration monitoring (e.g., Aldridge et al. 2009) or general assessments of stream ecosystem condition (e.g., EHMP 2008). Sudduth et al. (2011) found no difference in GPP or ER between four restored urban streams, four degraded streams, and four forested streams. Instead, they found that the primary driver of ER in their systems was temperature, which was strongly correlated with amount of impervious cover in the watershed.

Nutrient Dynamics

Water Quality – There are three common methods for evaluating nutrient dynamics in streams: nutrient uptake, nutrient budgets, and denitrification rates. Each provides different types of information but all are a reflection of water quality. Nutrient uptake metrics represent short term measurements and evaluates the rate of removal of dissolved nitrogen or phosphorus from the water column. Mechanisms of uptake in stream ecosystems include microbial and algal assimilation, microbial denitrification, physical adsorption, burial in the stream sediments, and volatilization (Bernot and Dodds 2005). Nutrient uptake measurements are useful because they can be performed over a short time period and they are done at the scale of entire reaches. However, because observation of net removal could be due to adsorption (e.g., of ammonia or phosphorus to sediments) or temporary storage in the sediments, uptake measurements are a better measure of how well a restored stream is 'functioning' with respect to nutrient processing than whether or not the restoration resulted in a net decline in the downstream flux of nutrients over time. Denitrification measurements in theory are much better because they provide information on the actual conversion of dissolved nitrate to harmless nitrogen gas; however, these measurements are technically difficult to make and interpret. As we discuss below, nutrient budget methods are time-consuming but provide the best measures of how effective a restoration project has been at reducing the downstream flux of nutrients (the same approach can also be applied to sediment flux).

Nutrient uptake is a direct, integrative measure of the functional capacity of stream ecosystems to remove nutrients (Stream Solute Workshop 1990). It is measured by adding a small amount of dissolved nutrients to a stream reach and taking water samples over time (usually ~2-6 hours) at multiple places downstream of the addition point to determine when nutrient concentrations return to background levels (for a full description see Hauer and Lamberti, 2006, Methods in Stream Ecology text). Return of nutrient concentrations to background levels indicates that all of the added nutrients have been used, presumably mostly by microbes and algae. Several physical and biological factors that influence nutrient uptake have been identified, including stream size (Alexander et al. 2000, Peterson et al. 2001, Wollheim et al. 2001), channel morphology and complexity (Grimm et al. 2005, Mulholland et al. 1997, Sweeney et al. 2004, Valett et al. 1996), organic matter availability (Meyer et al. 2005, Webster et al. 2000), stream metabolism (Hall and Tank 2003, Hoellein et al. 2007, Niyogi et al. 2004, Webster et al. 2003), and background nutrient concentrations (Dodds et al. 2002, Earl et al. 2006, Webster et al. 2003), all of which may be indirectly affected by land use (Hall et al. 2009).

Nutrient budgets are the best approach for determining if nutrient removal in a stream or a restoration reach is within the range of variability of least-disturbed or reference streams. This approach determines total nutrient removal (and/or long term storage) by measuring nutrient concentrations above and below a restoration site at regular intervals throughout the year and across the full range of discharges. Concentrations are converted to loads using the discharge data and the total annual flux of nitrogen that leaves the restored reach is subtracted from the total entering the reach to determine removal/retention. This method assumes there are no other major sources of water and nitrogen inputs along the reach (e.g., from tributaries or groundwater); if there are, then the amount of water and nitrogen in these inputs must be accounted for in the budget. This method is most often used for nitrogen because determining phosphorus fluxes requires measurement of both phosphorus adsorbed onto suspended sediments and dissolved phosphorus (although nitrogen in the form of ammonium will also sorb to sediments).

Denitrification rates are a measure of the permanent removal of N from the water via microbially-mediated reduction of nitrate to gaseous forms (N_2 and N_2O) under anaerobic conditions. Debris dams, organic-matter rich sediments, riparian zones, floodplains, and stream banks are all locations of potentially high N removal (Groffman et al. 2005), but rates vary with local conditions (e.g., soil characteristics (moisture content, grain size) and availability of nitrate and carbon; Baker and Vervier 2004, Mehnert et al. 2007, Vidon and Hill 2004). Denitrification capacity may be quantified in the laboratory using a denitrification enzyme activity (DEA) assay (Groffman et al. 1999, Smith and Tiedje 1979) or in the field using the 'push-pull' method (Groffman et al. 2006, Istok et al. 1997), both of which are applicable in a variety of terrestrial and aquatic systems. These methods have been used to estimate denitrification capacity in riparian and floodplain sediments (Addy et al. 2002, Kaushal et al. 2008, Kellogg et al. 2005, Klocker et al. 2009, Lowrance 1992), hyporheic zones (Clilverd et al. 2008), and shallow groundwater (Mehnert et al. 2007), as well as in-stream features (pools, riffles, gravel bars, debris dams; Groffman et al. 2005). Denitrification rates are both spatially and temporally variable, making scaling up of point-based measures to reach or whole stream levels difficult (Addy et al. 2002, Groffman et al. 2006). However, with replicate measurements made over dominant patch types and across seasons, actual or potential denitrification rates may provide valuable insights into the N removal capacity of an ecosystemand factors controlling N processing before and after the implementation of restoration projects.

Use of these three methods – uptake, budgets, denitrification rates – to evaluate restoration outcome is fairly recent. Bukaveckas (2007) used measures of N and P uptake, conducted in restored and unrestored reaches, to determine that a channel reconfiguration project in Kentucky was successful at improving in-stream nutrient processing (primarily as a result of flow velocity reduction). Aldridge et al. (2009) found increases in P uptake in response to a restoration project involving the addition of organic matter to an urban stream in Australia. Roberts et al. (2007) found rates of N uptake increased in response to a restoration experiment in which large woody debris was added to a stream in Georgia (Roberts et al. 2007). Sudduth et al (2011) found higher summer N uptake rates in four restored urban streams compared to forested streams but not compared to urban degraded streams; there were no differences among any streams in the winter. They provide strong evidence that the changes in nutrient uptake were related to removal of trees at the restoration sites – the open canopy led to increases in temperature and light which increased microbial uptake rates and also algal growth. They concluded that: "the major ecosystem impact of natural channel design restoration projects [in their restored streams] was to increase light and heat, which may stimulate nutrient uptake but is likely to have severe consequences for sensitive biota (Violin et al. 2011)." Klocker et al. (2009) used direct measures of nutrient uptake to determine that floodplain reconnection and the addition of step-pools in an urban stream in Maryland had no impact on N uptake. Nitrogen budget calculation methods were used by Filoso and Palmer (2011) on several restored Coastal Plain streams in Maryland. They found no evidence that restoration reduced the downstream flux of nitrogen except for a stream that had been 'restored' by converting it to a stream-wetland complex. Denitrification measurements on restored streams have been used by Kaushal et al. (2008) on an urban stream in Baltimore, Maryland; they found that banks that were reconnected to the floodplain (via bank re-grading) exhibited significantly higher rates of denitrification than unrestored, incised banks.

Summary of stream restoration metrics

Streams are highly dynamic in their natural state, which makes it difficult to assess restoration outcome over short time periods or with use of structural metrics that involve point in time measurements. Ideally, metrics should provide information on biophysical processes that are required to support biota and a healthy ecosystem. The most important metric to evaluate is discharge regime because of the dominant role that flow plays in determining channel dynamics, ecological processes, and biotic composition and abundance. Metrics such as biological diversity or presence/absence of particular taxa are excellent for evaluating stream *condition* but are poor metrics for evaluating whether or not a stream is on a trajectory toward recovery post-restoration. In addition to continuous discharge measurements, other process-based measures such as sediment flux, nutrient retention, and inputs of leaf litter and light are more valuable. However, it is critical to be fully cognizant of potential constraints to recovery such as impervious cover in the watershed.

Table 4-2. Summary of evidence for stream restoration metrics, including effort and cost

Metric	Effort	Cost	Strength of Evidence for Vital Habitat (VH) and Water Quality (WQ)
Hydrology, Geomorphology and Habitat			
Discharge regime	Moderate	Moderate	VH and WQ: The flow regime (magnitude, timing, and duration of discharges) is considered a "master variable" in streams because it exerts control over every ecological factor and, in concert with sediment inputs, controls channel form. Because it takes years to characterize the full regime, the magnitude and timing of high and low flows are the focus of recommended metrics. There is substantial evidence that these aspects of the flow regime influence vital habitat for invertebrates and may control water quality through effects on erosion, deposition, and functional attributes, such as rates of denitrification and decomposition.
Substrate characteristics	Low	Low	VH: The type and size of substrate particles influences biota including algae, macroinvertebrates, and fish. The % fine particles per unit area of streambed is particularly important since it is related to invertebrate and fish abundance and composition: at high levels it impedes burrowing, smothers eggs, and reduces the flux of oxygen into the streambed. WQ: Substrate also influences rates of ecological processes such as surface-water and hyporheic water interactions, and through its indirect effect on biota (especially microbes), stream metabolism. Increased silt and clay (% fines) has been correlated with increased abiotic P uptake.

Channel morphology	Moderate	Low	VH: Channel morphology has been correlated with macroinvertebrate diversity and composition in streams that are otherwise unimpaired (e.g., high WQ), though effects are species or group specific and highly scale-dependent. WQ: Channel morphology (especially channel complexity) has been proposed as a predictor for rates of processes such as litter retention and denitrification; however, to date few published studies have shown a strong relationship. Changes in channel geometry may be used to estimate erosion.
Flood plain accretion/ sedimentation rate	High	Low	WQ: Quantitative assessment of sedimentation rates provides a good metric for assessing progress in a project designed to reduce suspended sediment load and increase floodplain overflows; however, it is difficult to measure with a high degree of certainty. Spatial and temporal variability may make extrapolating to ecologically meaningful scales difficult.
Physical habitat assessments	Low	Low	VH: Habitat indices may or may not be correlated with biotic indices (e.g., macroinvertebrates, fish, periphyton) and are not an indicator of ecosystem function. Typically, poor habitat is not the cause of biological impairment in Chesapeake Bay streams.
Presence of large woody debris (LWD)	Low	Low	VH: LWD has been positively correlated to biotic metrics in some studies but not others. LWD addition leads to aggregation of fish but not necessarily increased fish production. LWD has been positively correlated to organic matter (OM) retention, though OM often does not increase following LWD additions. WQ: LWD is positively correlated to N and P uptake, and occasionally to sediment retention.

Biota - macroinvertebrates	Moderate	Moderate	VH and WQ: Macroinvertebrate community metrics have been correlated with water quality and habitat characteristics including temperature, conductivity, sedimentation rate, nutrients, metals, substrate type, and organic matter availability. Direction of effects varies by species or functional group, but high sedimentation usually lowers scores. Useful for monitoring outcomes, but not always useful as a diagnostic or adaptive management tool since responsiveness can vary widely; despite common perceptions, metrics are not reflective of all ecosystem functions.
Biota - fish	Moderate	Moderate	VH and WQ: Fish community metrics have been correlated with water quality and habitat characteristics including temperature, hydrology, nutrients, availability of basal resources, and substrate type. Direction of effects varies by species or functional group. Useful for monitoring outcomes, but not always useful as a diagnostic or adaptive management tool since responsiveness can vary widely; despite common perceptions, metrics are not reflective of all ecosystem functions. WQ: Fish community metrics are usually negatively correlated with suspended sediment concentrations.
Biota - periphyton	High	Moderate	VH and WQ: Periphyton community metrics have been correlated with water quality and habitat characteristics including nutrients, substrate size, and pH. Direction of effects varies by species or functional group. Useful for monitoring outcome but not always useful as a diagnostic or adaptive management tool since responsiveness can vary widely; despite common perceptions, metrics are not reflective of all ecosystem functions.
Riparian vegetation - % canopy cover	Low	Low	VH and WQ: Canopy cover is associated with the type and availability of basal resources (algae, leaf litter) and may influence food web structure through indirect effects on macroinvertebrates and fish. Increased canopy cover is positively correlated to rates of ecosystem respiration and negatively correlated to rates of primary production.

Riparian vegetation - % native vegetation	Moderate	Low	VH and WQ: Compared to the information on vegetation size and the extent of canopy cover, few studies have looked at the relationship between native vs. nonnative tree species and stream ecosystem structure or function. However, there are many studies throughout the world showing that the introduction of nonnative vegetation can dramatically influence stream ecosystems, especially if they are invasive and displace natives.
Physicochemical parameters (conductivity, DO, pH, temperature, metals, nutrients)	Low	Moderate	VH and WQ: Many physical and chemical parameters are correlated with biotic community metrics, though direction of effects varies by species or functional group; metrics are useful to provide context for other results. WQ: Provides indication of current water quality, but instantaneous measures do not capture variability that may include short periods of poor WQ that eliminate biota. Nutrient uptake is proportional to concentration, but % removal is lower at high concentrations (i.e., less efficient).
Benthic organic matter (BOM) availability	Moderate	Moderate	VH: If other factors are not limiting, BOM availability is positively correlated to macroinvertebrate survivorship, growth, abundance, and diversity in temperate forest streams. Relationships between BOM and whole-stream metabolic rates are unclear. WQ: BOM availability has been positively correlated to both N and P uptake.
Leaf litter decomposition rates	Moderate	Low	VH: Decomposition rates are an integrated measure of ecosystem function but are difficult to interpret; i.e., high rates may be due to excessive peak flows or to an abundance of shredding invertebrates. Requires other measures to interpret.
Whole-stream metabolism	Moderate	High	VH: Stream metabolism is an integrated measure of ecosystem function that provides an overall assessment of one aspect of stream health; both high and low rates are ‘normal’ depending on the context, thus, to interpret, rates must be compared to reference streams. WQ: Stream metabolism has been related to nutrient uptake in several studies.

Nutrient budgets	High	High	WQ: Since nutrient inputs to streams and nutrient uptake rates once the water is in the stream can vary tremendously seasonally and with respect to discharge, this is the best method for determining if a restoration project results in a greater net removal rate of nitrogen over time.
Nutrient uptake rate	High	High	WQ: Nutrient uptake is a direct, integrative measure of the functional capacity of streams to remove nutrients. Various factors influence the rates including flow, nutrients, oxygen levels, etc.
Denitrification rates	Moderate to High	High	WQ: Allows for estimation of the amount of N removed through denitrification pathways; high spatial and temporal variability may make extrapolating to ecologically meaningful scales difficult.

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