Water quality assessment

March 2022

Guidance Manual for Assessing the Quality of Minnesota Surface Waters for Determination of Impairment: 305(b) Report and 303(d) List

2022 Assessment and Listing Cycle

MINNESOTA POLLUTION CONTROL AGENCY

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Foreword

Minnesota is blessed with abundant water resources. Our lakes, rivers, and streams play a vital role in the state's economy and the rich quality of life residents and visitors enjoy. The health of Minnesota's environment and enormous opportunities for water-related recreation depend on good water quality.

Since the Clean Water Act became law in 1972, significant and often dramatic improvements in the water quality of Minnesota's surface waters have been accomplished. Notable examples include the Mississippi River below the Twin Cities, the Rainy River below International Falls, and the recent improvements to dissolved oxygen concentrations in the Minnesota River. Most of these gains can be attributed to vast improvements in domestic and industrial wastewater treatment.

In spite of these success stories, many Minnesota lakes and streams do not fully support beneficial uses such as swimming and fishing. The contribution of pollutants from nonpoint sources, from agriculture, construction and development sites, forestry, urban runoff, etc., is now the major reason that many of Minnesota's waters are considered impaired. The prevention and control of nonpoint source pollution remains one of Minnesota's greatest challenges.

The Minnesota Pollution Control Agency (MPCA) is charged under both federal and state law with protecting the water quality of Minnesota's lakes, rivers, streams, and wetlands. It is the responsibility of the MPCA to monitor Minnesota's water bodies, to assess water quality, and to report the results to the public. This task extends to documenting the water quality "success stories," as well as identifying those water bodies that still need improvement. MPCA is also working to better understand and address disproportionate impacts related to water quality. Some communities – particularly lower income communities or communities of color – may experience more impact when waters are polluted and uses are not supported. These communities may have less access to waters that are routinely safe for recreation; if they rely on locally caught fish as a large part of their diet, they have more exposure to pollutants in fish tissue.

This Guidance Manual was developed to help federal, tribal, state, and county staff, and the public in general, understand the water quality assessment process, and how Minnesota assesses water quality. The methodologies in this Guidance Manual have been refined in order to derive the most information, value, and benefit possible from available water quality data. The information created in the assessment process becomes the basis for evaluating the current status of Minnesota's water quality, identifying waters that are either impaired and need restoration or need further protection to prevent impairment, and tracking progress over time.

This Guidance Manual will be updated as assessment methods improve and as new pollution problems emerge that require assessment. Comments and suggestions from readers are encouraged and will be used to help improve the Guidance.

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Abbreviations, acronyms, and symbols

AUID	Assessment unit identification
BCC	Bioaccumulative Chemicals of Concern
BCG	Biological Condition Gradient
BOD	5-day Biological Oxygen Demand
CC	Chronic Criteria
chl-*a*	Chlorophyll-*a*, corrected for pheophytin
CS	Chronic Standard
CWA	Clean Water Act
DC	Domestic consumption
DNR	Minnesota Department of Natural Resources
DO	Dissolved oxygen
EPA	U.S. Environmental Protection Agency
EQuIS	Environmental Quality Information System
FCMP	Fish Contaminant Monitoring Program
GLI	Great Lakes Water Quality Initiative
HDS	Human Disturbance Score
HH-WQS	Human Health-based Water Quality Standards
IBI	Index of Biotic Integrity
IWM	Intensive watershed monitoring
LTRMP	Long Term Resource Monitoring Program
MDA	Minnesota Department of Agriculture
MDH	Minnesota Department of Health
MPCA	Minnesota Pollution Control Agency
NHD	National Hydrographic Data
PCB	Polychlorinated biphenyls
PJG	Professional Judgment Group
PFOS	Perfluorooctane sulfonate
QA/QC	Quality Assurance/Quality Control
RES	River Eutrophication Standards
RNR	River Nutrient Region
TALU	Tiered Aquatic Life Uses
TMDL	Total Maximum Daily Load
TP	Total Phosphorus
TSS	Total Suspended Solids
USGS	United States Geological Survey
WAT	Watershed Assessment Team
WQS	Water Quality Standards

Preface to the 2022 revision of the Guidance Manual

In this edition of the Guidance Manual for Assessing the Quality of Minnesota Surface Waters, the MPCA made these additions since the previous version published in 2020.

- Section V. B. 1. f. and Appendix G: Clarification of sestonic chlorophyll-a and periphyton data and water quality standards.
- Section VI. A. 2. b.: Inclusion of sample collection method required for mercury water quality assessments.
- Section VI. B. 2. c.: Inclusion of site-specific criteria in PFOS assessments.
- Section IX: New section on assessment of sulfate in wild rice waters.
- Section XI: Consolidation of delisting requirements for toxic parameters and clarification of delisting requirements for dissolved oxygen (DO) impairments.

I. Introduction

A. Background

Minnesota is blessed with abundant water resources. Our lakes, rivers, and streams play a vital role in the state's economy and the rich quality of life residents and visitors enjoy. The enormous opportunities for water related recreation these resources provide - such as aesthetic enjoyment, swimming, fishing, boating and canoeing - depend, to a great extent, on good water quality. Within Minnesota's borders lie the headwaters of three major continental watersheds: the Great Lakes/St. Lawrence River, the Mississippi River, and the Red River of the North/Hudson Bay Watersheds. Minnesotans have the privilege, and the huge responsibility, of living "upstream" of millions of downstream users of these major waterways. Minnesota's water resources include about 105,000 river miles, 4.5 million acres of lakes and reservoirs including approximately 1.4 million acres of Lake Superior in Minnesota, and about 9.3 million acres of wetlands.

The Minnesota Pollution Control Agency (MPCA) is charged under both federal and state law with the responsibility of protecting the water quality of Minnesota's lakes, rivers, streams, and wetlands. One goal of the MPCA is to preserve the existing high quality of water bodies that are meeting standards, so beneficial uses are maintained. However, too many surface waters receive enough pollutant loading from a variety of sources that they do not meet one or more water quality standards (WQS). If the extent of the violations of standards exceed the guidelines spelled out in this Guidance Manual (Guidance), those surface waters are considered to be "impaired." MPCA then works to improve the quality of impaired waters so WQS are met and beneficial uses are maintained and restored, where these uses are attainable.

B. About the TMDL list, Assessment and Listing Cycle, and Integrated Report

The federal Clean Water Act (CWA) requires states to adopt WQS to protect waters from pollution. These standards define how much of a pollutant can be in the water and still allow the water to meet its beneficial uses, such as drinking water, fishing, and swimming. WQS are the fundamental tools used to assess the quality of all surface waters. For more detailed information regarding standards, see https://www.pca.state.mn.us/water/water-quality-standards. States must monitor and assess the water quality of their waters to identify those that are "impaired", i.e., not fully supporting their beneficial uses. Section 303(d) of the CWA requires states to publish and update a list of impaired waters for which a Total Maximum Daily Load (TMDL) Study is needed. This list, known as the "303(d) List" or "TMDL List", is updated every two years via the assessment of water quality data and an extensive public participation process. The draft TMDL List is developed by the MPCA and submitted to the U.S. Environmental Protection Agency (EPA) for final approval. The two-year timeline for assembling and submitting the draft TMDL List is known as the "assessment and listing cycle." This Guidance has been prepared to reflect the 2022 Assessment and Listing Cycle.

Section 305(b) of the CWA requires states to submit a report on the status of all of their waters to help measure progress toward the national goals of fishable and swimmable waters. Also on a two-year cycle, Minnesota submits the comprehensive results of all assessments in an "Integrated Report". The latest assessment results submitted by states and tribes to fulfill CWA 305(b) requirements can be viewed at https://mywaterway.epa.gov. The Integrated Report includes Minnesota's Impaired Waters List – an accounting of all known impaired waters, not just those requiring TMDLs – and a narrative component with programmatic information about protection and restoration efforts. As part of the assessment process and the development of the Integrated Report, all waters for which sufficient data have been collected to allow a review are assigned to a category of impaired, unimpaired, or insufficient information to determine impairment status according to EPA-established categories (Appendix A). To view the MPCA's most recent 303(d) Impaired Waters List and Narrative Report see https://www.pca.state.mn.us/water/minnesotas-impaired-waters-list.

C. Monitoring and assessment approach

The MPCA conducts a variety of surface water condition monitoring activities focused on providing critical information to assess the condition of Minnesota's water resources. This information is also used to assess potential and actual threats to water quality and to evaluate the effectiveness of management activities taken to address impairments and other threats to water quality. Monitoring conducted by other local, state, and federal agencies; citizen monitoring; and remote sensing data are also used for this purpose. For more details on the MPCA's monitoring strategy, see https://www.pca.state.mn.us/water/water-quality-monitoring-strategy.

The MPCA's primary condition monitoring activities are organized around Minnesota's 80 major watersheds. The watershed monitoring approach involves intensive monitoring on a subset of major watersheds every year. The MPCA has implemented a schedule for intensively monitoring each major watershed once every 10 years, and the watershed outlets annually. These monitoring activities result in the identification of waters that are impaired and need restoration as well as waters that need further protection to prevent impairment. Monitoring is followed by TMDL prioritization and protection strategy development at the major watershed scale, and ongoing implementation. See https://www.pca.state.mn.us/water/watershed-approach-restoring-and-protecting-water-quality for a more in-depth discussion of the watershed approach and for a map of the 10-year watershed monitoring schedule. For information on TMDL priority rankings as they pertain to reporting to EPA, see Appendix B. An important feature of the watershed approach is the fact that restoration and protection planning and associated implementation will occur in all watersheds; the identification of an impaired status is not the key trigger for follow-on planning and implementation.

An annual assessment process has been designed to keep up with the monitoring work and reflect the more detailed monitoring data available in each watershed. The development of an annual assessment process has been critical to the MPCA's implementation of the overall watershed approach. With assessments taking place immediately following completion of IWM, the entire process of monitoring-assessment-restoration-protection can be completed within 10 years, at which time the watershed comes up for monitoring again as part of the next scheduled 10-year rotation. This allows clear assessment of progress towards meeting water quality goals. In addition, the revised process encourages earlier and more meaningful local involvement in assessment.

MPCA and Minnesota Department of Natural Resources (DNR) collaborate on the assessment of aquatic life in lakes utilizing a lake fish Index of Biotic Integrity (and a review of existing plant data). Sampling by DNR has been aligned so that aquatic life assessments are completed annually following the watershed monitoring approach.

For the purposes of fulfilling our monitoring and assessment objectives, large rivers are defined as large mainstem rivers that flow through multiple major watersheds and, therefore, were not satisfactorily represented within the watershed approach. In Minnesota, these rivers are monitored and assessed longitudinally on a rotating basis once every 10 years and include the St. Croix, Minnesota, Upper Mississippi, Red, and Rainy Rivers. The Lower Mississippi (below Upper St. Anthony Falls) also meets the definition of a large river but is treated separately due to ongoing interstate efforts to develop a consistent and comprehensive monitoring strategy to fulfill CWA objectives for interstate waters of the Mississippi River. For more on the MPCA's large river monitoring strategy, see https://www.pca.state.mn.us/water/large-river-monitoring.

Some monitoring, namely monitoring of toxic parameters, continues to occur on a statewide basis. Assessment of those parameters is done statewide every two years, to reflect the monitoring design. Watershed assessments focus primarily on the aquatic life and recreation beneficial uses. Statewide assessments focus primarily on aquatic consumption and aquatic life toxicity. Every two years the watershed and statewide assessment results are packaged together into the Impaired Waters List and Integrated Report.

For the 2022 Assessment and Listing Cycle, the watersheds and basins are:

Assessed in 2020	**Assessed in 2021**
Le Sueur River	Buffalo River
Little Fork River	Cedar River Basin
Mississippi River – Lake Pepin	Chippewa River
Root River	Lower St. Croix River
Sauk River	Mississippi River – St. Cloud
	St. Louis River
	Upper Red River

In 2021 for the 2022 Assessment and Listing Cycle, the following statewide assessments where performed:

- Nitrate in lakes and streams used as a source for drinking water.
- Pesticide and fish tissue contaminants in waters where data were available.
- Special assessment of sulfate in waters used for the production of wild rice.

While the MPCA's monitoring and assessment efforts primarily follow the major watershed schedule, interested parties are able to propose additional listings outside of the watershed schedule during the call for data or public notice of the draft Impaired Waters List. This proposal process accommodates instances when assessment and listing outside of the watershed schedule is necessary for a locally led initiative to move forward. To honor the watershed schedule and maintain the integrity of the systematic approach to monitoring/assessment, TMDL development, and implementation, any proposals for listing outside of the watershed schedule must 1) explain why moving forward with assessment is necessary prior to the comprehensive watershed assessment, 2) document how the efficiency and coordination that is lost by deviating from the watershed approach will be offset by a local benefit, and 3) demonstrate that the MPCA's assessment methods in this Guidance were followed for the monitoring, analysis, and comparison of the data against state standards. The MPCA reviews any such proposals and makes a determination regarding impairment and listing prior to submitting the draft list to EPA for approval.

II. Purpose and scope

A. About the Assessment Guidance

The purpose of this Guidance is to define required data and information and lay out the criteria by which water bodies are assessed to determine if beneficial uses are supported.

The scope of this Guidance includes methods for assessing surface waters for the following beneficial uses:

- Aquatic life (toxicity-based standards, conventional pollutants, biological indicators).
- Drinking water and aquatic consumption (human health-based standards).
- Aquatic consumption (fish-tissue and wildlife-based standards).
- Aquatic recreation (*Escherichia coli* – *E. coli* – bacteria, eutrophication).
- Limited value resource waters (toxicity-based standards, bacteria, conventional pollutants).

B. Disclaimers and future changes to the Guidance

To people not involved with conducting water quality assessments, the determination of an impaired condition would seem to be a straightforward process: waters either meet standards or do not. However, the assessment process is complex and it includes a certain amount of uncertainty.

The MPCA must consider many different types and sources of data, different categories of pollutants, different uses of surface waters, the variability in natural systems, and many other factors. The goal of this Guidance is to describe the assessment methods accurately and completely, and to make the assessment process as clear and understandable as possible. Nevertheless, questions about the assessment process will invariably arise that the Guidance fails to answer. Readers are encouraged to access the many resources listed in Section XII, including MPCA staff, for additional information. Two MPCA products that may be especially useful are the Volunteer Surface Water Monitoring Guide (MPCA 2003) (http://www.pca.state.mn.us/water/monitoring-guide.html) and the Surface Water Data website (http://cf.pca.state.mn.us/water/watershedweb/wdip/search_more.cfm). The Monitoring Guide provides information on planning a monitoring program, as well as data quality and management. The Surface Water Data website allows Minnesotans to access environmental data on surface waters statewide.

This Guidance does not affect the rights and administrative procedures available to all affected or interested parties. The Guidance is not part of any water quality rule – it does not have the force of law. It serves to guide the interpretation and application of current WQS that are in water quality rules. If any party feels that an MPCA decision based on the Guidance is not supported by the facts, or they have any issue related to the MPCA's use of the Guidance, that party can comment on the MPCA's actions in the following ways:

- Directly contact MPCA staff, management, or the Commissioner, orally or in writing.
- Request a contested case hearing if the issue involves an MPCA permit action, or any other MPCA action for which a contested case hearing is an appropriate forum to resolve the concern.
- Challenge the MPCA action in the appropriate legal jurisdiction.

The MPCA updates this Guidance every two years in conjunction with the current EPA-mandated schedule for preparation of both the 305(b) Integrated Report and the 303(d) List. The MPCA involves the public when major changes to the Guidance are being considered and invites the public to comment on this Guidance on the same schedule as the 303(d) Impaired Waters List.

C. Other standards

Other toxic or conventional pollutants that are found to exceed WQS will be assessed following equivalent methodologies discussed in this Guidance, depending on the type of pollutant. Methodologies will be developed and included in this document as new pollutants are added to the assessment process.

III. Steps in the assessment process

As noted in the Introduction, the MPCA maintains a watershed-based monitoring, assessment, and restoration/protection schedule. Minnesota's Water Quality Monitoring Strategy 2021-2031 discusses this process combines computerized data analysis, expert review, and internal and external partner input, ensuring that all available data and information is used to make appropriate assessment decisions (MPCA 2021). Assessments of use-support in Minnesota are made for individual water bodies. The water body unit used for stream reaches, lakes, and wetlands is called the "assessment unit" (AUID). See Section IV for details.

It also stressed and engrained the importance of quality assurance and quality control at every step in the process. Further detail on the specific steps in the process is included below. A note should be made that the aquatic consumption (fish) assessment at this time utilizes only the first two steps in the process.

A. Data compilation

The initial step in the process is a computerized screening that identifies monitoring results collected on AUIDs over the appropriate period of record and compares each data point to water quality criteria, summarizes the number of data points that exceed the criteria, the total number of data points, and the number of years of data. This step produces a parameter-specific summarization (e.g., dissolved oxygen (DO), Fish IBI, and *E. coli*) and is maintained in MPCA's assessment database. For more information on the sources of data that the MPCA uses, see Appendix C.

B. Desktop assessment

The desktop assessment involves a review of data and summaries for water bodies within a specific major watershed, or 8-digit hydrologic unit code watershed (HUC-8). It is performed by resource-specific staff, i.e., water quality staff review chemistry data, biologists review stream biological data, DNR staff review lake biology, and specialists review toxic parameters such as pesticides and nitrate. Staff ascertain the quality of the dataset (temporal and spatial completeness, etc.) and consider multiple lines of evidence including but not limited to flow conditions, precipitation, land use, and habitat. The results of which are a recommendations as to whether data show the parameters are is meeting or exceeding the appropriate standards. During this process, any candidates for recategorization (a move of an impairment out of Category 5, see Appendix A) are identified and work begins to justify those changes to the Impaired Waters List.

C. Watershed Assessment Team (WAT)

The WAT includes desktop assessors, regional watershed project managers, stressor identification staff, and other state agency personnel involved in the HUC-8 assessments. Invites are also extended to Tribal water quality personnel for HUC-8 watersheds that include waters wholly or partially within Tribal boundaries. The WAT meets to review each AUID in the watershed, considering comments and parameter-level evaluations from the desktop assessment as well as supplemental information, to reach an overall use-support decision. Delisting and natural background candidates may also be identified at this time.

D. Professional Judgment Group (PJG)

The PJG is comprised of WAT and external parties (local data collectors, local government units, tribes, etc.), as determined by the MPCA regional watershed project manager. This group meets to discuss the results of the WAT meeting for a specific HUC-8. Prior to the PJG meeting, the results of the WAT meeting are distributed to all invitees, including parameter-level evaluations, overall use-support recommendations and all decision comments. Invitees are asked to identify AUIDs they wish to discuss; an agenda is developed based on these submissions. The format of this meeting is an overview of the process, a general discussion of the watershed and major subwatersheds, and a review of requested AUIDs, recategorization candidates. It does not include an exhaustive review of each AUID. The PJG meetings result in final use-support determinations for the Integrated Report. If applicable, border states are consulted and reasons for any discrepancies in assessment determination between Minnesota and the specific border state are documented.

The analyses and recommendations for each AUID are documented in a database and archived following the completion of the assessments. Throughout the annual assessment process, care is taken to maintain consistency among the HUC-8 assessment meetings and decisions. This is accomplished via internal training and quality control, and oversight and guidance provided by a technical team and a management team charged with ensuring quality data analysis and consistency among watershed assessment discussions and decisions.

IV. General aspects of data assessment

A. Delineation of reaches, lakes, and wetlands

The MPCA uses the 1:24,000 scale high resolutions National Hydrography Dataset (NHD) to create geospatial data to represent stream and lake assessment units. All of our assessment units are indexed to the NHD, or have had custom shapes created for addition to the NHD. The high resolution NHD was created from 1:24,000 scale United States Geological Survey Digital Line Graphs and DNR stream and lake data.

Each water body is identified by a unique water body identifier code called an assessment unit identification or AUID. For streams, the code is comprised of the United States Geological Survey (USGS) 8-digit sub-basin code plus a three-character code that is unique within each sub-basin. It is for these specific reaches that the data are evaluated for potential use impairment. A stream assessment unit usually extends from one significant tributary to another or from the headwaters to the first significant tributary. They are and is typically less than 20 miles in length. Main-stem large rivers utilize hydrologic unit boundaries (10-digit HUC) as the initial assessment unit. A stream or river reach may be further divided into two or more assessment units when there is a change in the use classification (as defined in Minn. R. 7050, https://www.revisor.mn.gov/rules/7050/), or when there is a significant morphological feature such as a dam, or a lake within the river.

The DNR's Protected Waters Inventory is the source for lake and wetland identifiers. DNR uses an 8-digit identifier for water bodies, consisting of a 2-digit prefix that represents county, 4-digit number identifying a lake, and a 2-digit suffix that represents either a whole lake (-00) or representing a specific bay of a lake (-01, -02, etc.). This 8-digit identifier is used by MPCA to represent an assessment unit for lakes and wetlands. The MPCA reviews waters for wetland determination as needed during the assessment process using the criteria identified in Appendix D. Water bodies determined to be wetlands will not be assessed using the eutrophication factors discussed in Section VIII. C.

For the purposes for identifying water bodies as either wholly or partially within federally recognized Indian reservations, the MPCA uses the U.S. Census Bureau's spatial data on American Indian/Alaska Native Areas/Hawaiian Home Lands. Waters that flow through, or are completely within, reservation boundaries receive a special notation in Minnesota's Impaired Waters List. Those lakes and streams that serve as a boundary between state land and reservation land do not receive notation and are treated, in assessment and listing, the same as border waters between neighboring states. The U.S. Census Bureau's data are public and available at https://www.census.gov/cgi-bin/geo/shapefiles/index.php. For more information on the MPCA's approach for assessing and communicating the quality of waters that occur partially or wholly within federally recognized Indian Reservations, see Appendix E.

B. Period of record

The MPCA uses data collected over the most recent 10-year period for all the water quality assessments. Years of record are based on the USGS water year, October 1 of one year through September 30 of the following year. It is preferable to split the year in the fall, when hydrological conditions are usually stable, than to use calendar years. The MPCA uses the 10 year period in its assessments because this period is long enough to provide reasonable assurance that the data has been collected over a range of weather and flow conditions and that all seasons are adequately represented. From a practical standpoint, the 10-year period means there is a better chance of meeting the minimum data requirements. A full 10 years of data are not required to make an assessment.

In accordance with Minn. Stat. 114D.25, Subd. 6., the MPCA must take into consideration recent relevant pollution reductions resulting from controls on municipal point sources and nonpoint sources. In practice, this means that, if MPCA is aware of projects or facility changes that would result in a measurable improvement in the receiving water quality, the MPCA will consider these improvements in its assessment decision-making. Depending on the potential impact to water quality realized by these improvements or changes, the MPCA may:

1) Base its assessment decision solely on data collected post-project(s)/change(s),
2) Make its assessment decision by placing more weight on data collected post-project(s)/change(s); or
3) Defer an assessment decision altogether until sufficient post-project(s)/change(s) data can be obtained.

C. Uncertainty in water quality assessments

The MPCA is cognizant of the hazards of making assessments with limited data. One benefit of the watershed monitoring approach is that it provides a robust dataset for assessment. The selection of the minimum data requirements for water quality assessment is clearly a compromise between the need to assess as many water bodies as possible and the importance of minimizing the probability of making an erroneous assessment. The methods described in this Guidance deal with this problem in a variety of ways, depending on the pollutant category. Nonetheless, even with relatively robust datasets, some level of uncertainty is part of every analysis of water quality data. There is always a chance that a water body will be assessed as impaired when in fact it is not, or assessed as un-impaired when in fact it is. The number of data points the MPCA requires as a minimum for water quality assessments is small in the context of statistical analyses of uncertainty. The approach used by the MPCA to make impairment decisions, which is a screening of the data using the impairment thresholds, followed by a review by professionals, makes the best use of limited data. This is the approach recommended by the EPA.

Essentially all assessments are subject to review by a team of professional water quality experts (see previous section). Review of the data by professionals is an important part of minimizing erroneous impairment determinations, and is required whether statistical tests of data uncertainty are used or not. The possible erroneous placement of a water body on the 303(d) List is a concern because of the regulatory and monetary implications of 303(d) listing; not placing a water body on the list misses opportunities for restoration and improvement. It has been the experience of the MPCA that very few water bodies have been incorrectly determined to be impaired.

When the professional review of data collected for a lake or stream finds conflicting or inadequate information to make a confident assessment, and more data could resolve the need, notes are recorded in the assessment database. Subsequent discussions with monitoring programs occur to determine who is responsible for additional sampling and when it can be completed.

D. Data sources and quality

Data for assessments are queried primarily from MPCA's water quality data management system, EQuIS (Environmental Quality Information System); a limited amount of data from outside that system is also included in the process. However, to allow for the external data to be included in the process, it must be submitted to MPCA in time for incorporation into the assessment tables; this date is announced via a call for data and is typically November 1 prior to the start of the assessments.

The data used in assessment decisions must be of reliable quality and QA/QC protocols must be carefully followed for each step along the way – from field sampling to lab analysis to data management – in order to reduce the introduction of errors. Monitoring and data management at the MPCA are performed in accordance with the requirements specified in a Quality Management Plan approved by the EPA and available for review on the MPCA website at https://www.pca.state.mn.us/data/mpca-quality-system. For more information on data sources see Appendix C.

E. Dataset quality and parameter-level evaluation

As noted previously, a key step in the assessment process is to determine if individual parameters meet or exceed their criteria (numeric or narrative standards) or have insufficient data to make that determination. In addition to this comparison against standards, the evaluator also makes a determination of the confidence of the parameter assessment, assigning a low, medium, or high quality rating. These results are stored in the assessment database and used in the WAT reviews and PJG meetings, with supporting information, to make the final use-support determinations.

For some parameters, the parameter-level evaluation is equivalent to the final use assessment decision (e.g., *E. coli* bacteria). For other parameters (e.g., conventional chemistry and biota), the parameter-level evaluations are then used in conjunction with supporting data, including consideration of dataset quality, to make a final use-support determination. This will be discussed further in specific sections that.

To assist in parameter-level evaluations, MPCA has developed guidance for technical staff to use in their analyses (Table 1). The 10% and 25% exceedance frequencies referenced in Table 1 for conventional pollutants are based on EPA guidance (EPA 1997) and have been used by the MPCA in assessments for many years. These thresholds are appropriate for the conventional category of pollutants for several reasons, including that none are considered “toxic” (or bioaccumulative), and all are subject to periodic “exceedances” because of natural causes. For example, total suspended solids (TSS) levels typically increase in streams after a rain event even in relatively undisturbed parts of the state, and DO can drop below the standard in low gradient rivers and streams for reasons other than pollution, such as the flow of a stream through extensive wetland complexes. These potential pollutants are also natural characteristics of surface waters, the fluctuations of which aquatic organisms have adapted to cope with over time. The existence and extent of natural exceedances are considered during the assessment process.

The dataset quality rating and notes about the parameter-level evaluation are recorded for use by the WAT and PJG in making the use-support assessment. The technical staff that completed the parameter-level evaluations participates in the WAT and PJG meetings.

Table 1. Guidelines for parameter-level evaluations of conventional pollutants.[1]

Assessment	Frequency of exceedances	Magnitude of exceedances	Duration of exceedances	Timing of exceedances[2]
Water chemistry parameter indicating unimpaired or supporting conditions	Less than 10% exceedances of chronic standard	Exceedances generally within 10% of water quality criteria	Continuous data or extensive grab sample data set indicates no or few instances of prolonged exceedance	Exceedances only occurring during extreme events such as 100 year flood or severe drought conditions
Water chemistry parameter indicating potential impairment	Between 10 – 25% exceedances of chronic standard	Exceedances generally greater than 10% but less than 25% of water quality criteria	Continuous data or extensive grab sample data set indicates some instances of prolonged exceedance	Exceedances only occurring during periods in which they are most likely to occur (e.g., before 9 am, low flow conditions, storm events, etc.); not counting extreme events above
Water chemistry parameter indicating potential for severe impairment	Greater than 25% exceedances of chronic standard	Exceedances generally greater than 25% of water quality criteria	Continuous data or extensive grab sample data set indicates chronic exceedance or many instances of prolonged exceedance	Exceedances occurring during periods (seasonal or daily cycle) in which they typically do not occur in addition to occurring in periods in which they are most likely to occur.

[1]Most parameters will have data sets that only allow frequency and magnitude to be evaluated. When sufficient data exist (e.g., continuous monitoring or extensive grab samples) or appropriate ancillary data (e.g., flow, precipitation) are accessible, duration or timing of exceedances may also be considered in the evaluation. The parameter-level evaluation requires best professional judgment to integrate information across all applicable columns.

[2]Based on evaluation of available flow data and/or precipitation records as well as observations made by monitoring staff.

F. Reporting

MPCA reports the results of the assessments in a number of different formats, in watershed assessment reports, and in integrated reporting to EPA. A brief description of each is below.

1. Watershed Monitoring and Assessment Report

Results of the assessments are compiled in a HUC-8 watershed monitoring and assessment report following the assessment determinations. AUIDs are discussed by subwatersheds and overall water quality conditions, potential stressors, and protection areas are identified. These documents inform the restoration (TMDL) and protection (WRAPS) strategies that are developed by the agency. An example of a watershed assessment report can be found at https://www.pca.state.mn.us/sites/default/files/wq-ws3-09030005b.pdf.

2. Integrated reporting

The results of the assessments are reported as directed by guidance from EPA. The assessment data are loaded into EPA's ATTAINS database and are made available at https://mywaterway.epa.gov. Categories and subcategories used to classify each assessment unit can be found in Appendix A. Impaired use/pollutant combinations without approved TMDL plans make up the 303(d) List. In conjunction with the assessment data, a narrative report to the U.S. Congress as required by section 305(b) of the CWA is developed; this can be found at https://www.pca.state.mn.us/water/minnesotas-impaired-waters-list. An Integrated Report consisting of the narrative report, ATTTAINS data, a 303(d) List and NHD indexed geospatial data are completed and submitted to EPA by April 1 every even year.

V. Protection of aquatic life

A. Pollutants with aquatic life toxicity-based water quality standards

Protection of "aquatic life" with applicable Class 2 chronic standards means protection of the aquatic community from the direct harmful effects of toxic substances, and protection of human and wildlife consumers of fish or other aquatic organisms. This section of the Guidance deals with the former, the assessment of water quality for pollutants that have aquatic life toxicity-based chronic standards (CS) and acute or maximum standards (MS) that are always aquatic life toxicity-based. These standards are identified in Minn. R. 7050.0222 by the abbreviation, "Tox," and by column headings, "Aquatic Life Chronic Standards or Maximum Standards," in Minn. R. 7052.0100. These numeric standards are applied based on one-day average pollutant concentrations for the MS and four-day average concentrations for the CS.

Surface waters are assessed to determine if they are of a quality needed to support the aquatic community that would be found in the water body under natural conditions. In general, two types of data are used in assessments: water chemistry data and biological data. Computer-generated summaries based on chemistry data and biological data are both considered, along with data quality indicators, in aquatic life use-support determinations. Aquatic life use-support determinations are completed for all parameters/indices below for streams and for specific parameters/indices as noted for lakes and wetlands.

1. Toxic pollutants

The pollutants that have aquatic life toxicity-based standards most often included in MPCA water quality assessments are briefly discussed. Pollutants other than those mentioned here may be assessed also, as data allow.

a) Trace metals

Trace metals that have CSs to prevent toxicity to aquatic organisms and are used in water quality assessments include aluminum, cadmium, chromium III, chromium VI, copper, lead, nickel, selenium, silver, and zinc. Antimony, arsenic, cobalt, mercury, and thallium are discussed in Chapter VI because they have human health-based standards.

Minn. R. 7050 and 7052 include numeric standards for trace metals both in terms of "total" metal and, through conversion factors, "dissolved" metal. The use of dissolved metal standards is based on evidence that the dissolved analysis is generally a better estimate of the toxic fraction of metals in most water bodies, and it is EPA policy that metal standards should be in the form of dissolved metal (EPA 1993). The exception to this is aluminum. In recent years, additional research has demonstrated that the total fraction of aluminum is a better estimate of the toxic fraction. EPA has recently updated the aluminum criteria value, and has based it on total aluminum, rather than dissolved, reflecting the updated science (EPA 2018). Total and dissolved metal data will be used in the assessments. However, with the exception of aluminum, total metal data can be used to show that concentrations are less than (and thus meet) dissolved metal WQS, while total metal data cannot be used to indicate impairment.

The CSs for cadmium, chromium III, copper, lead, nickel, and zinc vary with ambient total hardness. Thus, the standards for these metals are in the form of formulas that reflect the hardness/toxicity relationship. Each measured value for a hardness-dependent metal is compared to an individually calculated standard based on the hardness at near the same time and place the metal sample was taken. If the measured hardness is above 400 mg/L, a maximum hardness cap of 400 mg/L is used to calculate the standard. If the measured hardness is below 50 mg/L, a minimum hardness value of 50 mg/L is used to calculate the standard.

Figure 1. Use of trace metals data for total metals standards.

Chronic Standard (CS) for Trace Metal (total)

Aquatic life toxicity-based
(Except for aluminum)

Convert CS to dissolved CS

Multiply total CS by adjustment
factor in Minn. R. 7050.0222, subp. 9

If factor <1.0
(dissolved CS
is < total CS)
(Adjustment factor is never > 1.0)

If factor =1.0, or no factor listed,
then factor = 1.0. total and
dissolved CS are equal

Result is dissolved CS

Compare **dissolved** CS to **dissolved** ambient data (filtered sample)

Human health-based
(And aluminum for aquatic life)

No conversion to dissolved CS

Compare CS to total [unfiltered]
analysis of ambient water

Hypothetical example: Total Copper CS = 15 µg/L @ a hardness of 200 mg/L
Total CS = 15 µg/L, aquatic life toxicity-based; factor = 0.960;
Dissolved CS = 14.4 µg/L (15 µg/L X 0.960)
Therefore, compare the 14.4 µg/L dissolved CS to the dissolved ambient copper analysis to assess for compliance with WQS.

b) Un-ionized ammonia

Ammonia at elevated levels in the un-ionized form (NH_3) is toxic to aquatic life. The chronic un-ionized ammonia standards are shown below:

- Class 2A. 0.016 mg/L un-ionized ammonia
- Class 2Bd, B, C, D. 0.04 mg/L un-ionized ammonia

The fraction of total ammonia in the un-ionized form in water is dependent on ambient pH and temperature. Therefore, pH and temperature as well as total ammonia must be measured at the same time and place to determine the un-ionized ammonia concentration.

c) Chloride

Besides being a general indicator of human impacts on water quality, high levels of chloride can harm aquatic organisms by interfering with the organism's osmoregulatory capabilities. The Class 2 CS for chloride is 230 mg/L.

d) Pesticides

The Minnesota Department of Agriculture (MDA) conducts extensive pesticide monitoring in surface waters and submits all data to the MPCA for assessments. At present, the MPCA has Class 2 chronic and maximum aquatic life standards for acetochlor, alachlor, atrazine (including degradates), chlorpyrifos, metolachlor, and parathion.

2. Data requirements and determination of impaired condition

Exceedances of standards for toxic pollutants are evaluated over consecutive three-year periods (see Table 2). Two or more exceedances of the CS in three years is considered an impairment. One exceedance of the MS is considered an impairment.

Aquatic life toxicity-based CS are written as four-day average concentrations. In some cases, pollutant concentrations can be quite variable over such periods, depending on factors such as the type and size of the water body, weather and flow conditions, and the source and nature of the pollutant. For example, chloride concentrations in lakes, streams, and wetlands are relatively stable during low flow conditions over a four-day period, while pesticide concentrations in small streams during storm events can vary greatly in that same amount of time.

Because the CSs are expressed as four-day averages, care must be taken to ensure that the water quality measurements used in assessments provide an adequate representation of pollutant concentrations over the relevant time period. When concentrations are judged to be relatively stable over the four-day period in question, single samples can be sufficient. When concentrations are more variable, multiple samples or time-weighted composite samples are can be used in order to calculate a sufficiently accurate average concentration. Flow-weighted composite samples are taken with the purpose of calculating average concentrations by volume rather than by time, and can be very difficult to interpret in assessment contexts and are therefore not used.

If more than one sample was taken within a four-day period for flowing waters the values are averaged, (usually an arithmetic mean is appropriate) and the four-day average is counted as one value in the assessment. This includes multiple samples in four days at one station or multiple stations along an assessment unit. For lakes, depth of sample must be taken into consideration, as concentrations may change with depth (i.e., chloride often increases with depth). Within the four-day period, samples will typically be averaged as follows: those samples collected at depths of 2 meters or less (including both grab samples and 0-2 meter integrated samples), those at maximum depth (defined as the deepest two meters of the water column), and the mid-depth values (taken between 2 meters from the surface and the maximum depth). As with flowing waters, this averaging applies to both samples at a single station or samples collected at multiple stations along the assessment unit. Each depth will be compared against the CS. If any four-day average, regardless of depth, exceeds the standard, it will count as a single exceedance for the water body (e.g., the surface average may meet the standard, while the average at 12 meters may exceed the standard – for that four-day period, a single exceedance will be counted).

The necessary number and type of samples can vary considerably from one situation to another and the determination of adequacy for the purpose of assessment will necessarily involve considerable professional judgment. It should be noted that because impairment can result from only one or two exceedances, a designation of meeting the standard generally requires extensive monitoring during times when exceedances are most likely to occur.

Table 2. Summary of data requirements and exceedance thresholds for assessment of pollutants with aquatic life toxicity-based standards.

Period of record	Use-support or listing category	
Most recent 10 years	No more than one exceedance of the Chronic Standard in three years, and no exceedances of the Maximum Standard: Not listed	Two or more exceedances of the Chronic Standard in three years, or one or more exceedances of the Maximum Standard: Listed

B. Conventional pollutants and biological indicators

Conventional pollutants or water quality characteristics most often included in MPCA water quality assessments are DO, pH, temperature, sediment and river eutrophication. Sediment is measured directly through TSS concentrations or estimated from Secchi tube measurements. River eutrophication consists of a causative variable (TP) and response variables indicating eutrophication. Biological indicators (fish and invertebrates in streams and fish in lakes) are currently evaluated in MPCA assessments.

Data summaries based on chemistry data and biological data are both considered, along with data quality indicators and supporting information, in aquatic life use-support determinations. Not all data types are available for all AUIDs, and not all datasets agree. The following paragraphs describe the parameter-level data that inform aquatic life use-support determinations and the process for evaluating the parameter-level and supporting data to make such decisions.

1. Pollutant or water quality characteristic

The conventional pollutants most often included in MPCA water quality assessments are briefly described. Pollutants other than those mentioned here may be assessed also, as data allow.

a) Low dissolved oxygen (DO)

DO is required for essentially all aquatic organisms to live. When DO drops below acceptable levels, desirable aquatic organisms, such as fish, can be harmed or killed. DO standards differ depending on the use class of the water (Minn. R. 7050.0222):

- Class 2A: Not less than 7 mg/L as a daily minimum.
- Class 2Bd, 2B: Not less than 5 mg/L as a daily minimum.
- Class 2D: Maintain background.
- Class 7: Not less than 1 mg/L as a daily average, provided that measurable concentrations are present at all times.

The standard for DO is expressed in terms of daily minimums and concentrations generally follow a diurnal cycle with concentrations increasing during the day and decreasing overnight. Consequently, measurements in open-water months (April through November) should be made before 9:00 a.m.

A stream is considered to exceed or not meet the standard for DO if 1) more than 10% of the “suitable” (i.e., taken before 9:00 a.m.) May through September measurements violate the standard and there are at least three such violations, or 2) more than 10% of the total May through September measurements violate the standard and there are at least three such violations, or 3) more than 10% of the total annual measurements violate the standard and there are at least three such violations.

Because the underlying criterion defines that WQS can be exceeded no more than 10% of the relevant time, it is usually essential that measurements are a representative sample of overall water quality and are not biased towards certain types of conditions, such as storm events, or certain times of the year. The relevant time generally refers not to the entire year but rather to the usual water quality monitoring portion of the year. The requirement of a violation rate of more than 10% helps ensure that the measured data set is sufficiently large to provide an adequate picture of overall conditions.

In spite of the significant water quality improvements that have resulted from application of the DO standard, the current standard is not necessarily appropriate for all streams. Some low-gradient, heavily wetland-influenced streams may never meet the current DO standard of 5 mg/L, even though pollutant sources and anthropogenic influences are insignificant or even non-existent. In such cases, the current DO standard is not a useful indicator of the health of the water.

Until the DO standard is refined to fit such situations, the following will apply:

- AUIDs where all monitoring sites have wetland characteristics significant enough to preclude the use of the current DO standard as well as current biological criteria will be designated as “not assessable” for aquatic life. The following statement will be used in the documentation: *Not assessed; the waterbody exhibits prevailing wetland characteristics. Assessment is deferred pending refinement of the assessment criteria or reclassification of the waterbody.* Where appropriate, some such waters will subsequently be moved into class 2D during the use-attainability review of the watershed.
- AUIDs where all monitoring sites have wetland influences significant enough to preclude the use of the current DO standard but which are assessable using biological criteria will be designated as “not assessable” for DO. The following statement will be used in the documentation: *Not assessed for dissolved oxygen; the current standard of 5.0 mg/L is not a reliable indicator of the health of this type of heavily wetland-influenced stream. Assessment for dissolved oxygen is deferred pending refinement of the assessment criteria.* (Individual monitoring sites within AUIDs can likewise be determined to be not assessable for DO because of wetland influences.)

A designation of meeting the standard for DO generally requires at least 20 suitable measurements from a set of monitoring data that give a representative, unbiased picture of DO levels over at least two different years. Continuous data, taken at 15- or 30-minute intervals will also be considered for assessment. However, if it is determined that the data set adequately targets periods and conditions when DO exceedances are most likely to occur, a smaller number of measurements may suffice for a determination of meeting the standard.

b) pH

The pH of water is a measure of the degree of its acid or alkaline reaction. The applicable pH standard for most Class 2 waters is a minimum of 6.5 and a maximum of 9.0, based on the most stringent of the standards for the multiple applicable beneficial uses. pH values that are outside the range of the standard because of natural causes are not considered violations.

Data are compared to the pH WQS for aquatic life use, where the standard for most Class 2 waters is 6.5 – 9.0. Different pH standards for aquatic life use apply to trout streams (6.5 – 8.5). A stream is considered to exceed the standard for pH if 1) the standard is violated more than 10% of the days as determined from a data set that represents unbiased conditions and 2) there are at least three measurements violate the standard.

A stream is considered to meet the standard for pH if the standard is met at least 90% of the days of the monitoring season. A designation of meeting the standard for pH generally requires at least 20 suitable measurements from a data set that gives an unbiased representation of conditions over at least two different years.

c) Total suspended solids (TSS)

TSS consists of soil particles, algae, and other materials that are suspended in water and cause a lack of clarity. Excessive TSS can harm aquatic life, degrade aesthetic and recreational qualities, and make water more expensive to treat for drinking.

Transparency values, as measured by Secchi tubes (S-tube), reliably predict TSS and can serve as surrogates. While TSS measurements themselves are generally preferred, datasets for S-tube are often more robust, and their relative strength will be considered in assessments.

Because S-tube measurements are not perfect surrogates, however, their use involves a margin of safety. Therefore, the S-tube surrogate thresholds for determining if a stream exceeds the TSS standard are different than for determining if a stream meets the standard (Table 3).

Table 3. Minnesota's TSS (mg/L), S-tube (cm) and site-specific standards for specifically named river reaches.

Region or River	TSS	S-tube Exceeds	S-tube Meets
All Class 2A Waters	10	55	95
Northern River Nutrient Region as Modified for TSS	15	40	55
Central River Nutrient Region as Modified for TSS	30	25	35
Southern River Nutrient Region as Modified for TSS	65	10	15
Red River Mainstem – Headwaters to Border	100	5	10
(Assessment season for above waters is April through September)			
Lower Mississippi River Mainstem – Pools 2 through 4	32		
Lower Mississippi River Mainstem below Lake Pepin	30		
(Assessment season for Lower Mississippi is June through September)			

Details regarding River Nutrient Region boundaries and assignments as adapted for application of the Minnesota TSS water quality standards can be found in Heiskary and Parson (2013) at https://www.pca.state.mn.us/sites/default/files/wq-s6-18.pdf, including a statewide map in Figure 4.

A stream is considered to exceed the standard for TSS/S-tube if 1) the standard is violated more than 10% of the days of the assessment season (April through September) as determined from a data set that gives an unbiased representation of conditions over the assessment season, and 2) least three measurements violate the standard. The Lower Mississippi River TSS standard is not met if summer

(June through September) average concentrations exceed the standard in more than half of the summers.

A stream is considered to meet the standard for TSS/S-tube if the standard is met at least 90% of the days of the assessment season. A designation of meeting the standard for TSS/S-tube generally requires at least 20 suitable measurements from a data set that gives an unbiased representation of conditions over at least two different years. However, if it is determined that the data set adequately targets periods and conditions when exceedances are most likely to occur, a smaller number of measurements may suffice. The Lower Mississippi River TSS standard is met if summer average concentrations do not exceed the standard in more than half of the summers.

S-tube measurements that fall between the two relevant surrogate values are considered to be indeterminate in exceeding or meeting the TSS standard. If a stream satisfies neither the criterion for exceeding the standard nor the criterion for meeting the standard, the stream is considered to have insufficient information regarding TSS levels.

d) Temperature

High water temperatures, or rapid elevations of temperature, can be detrimental to fish. Cold water fish such as trout are particularly intolerant of high temperatures. The temperature standard for Class 2A cold water sport fish is a narrative statement of "no material increase." Examples of demonstrating a "material increase" include temperature data showing a statistically significant increase when measured upstream and downstream of a stream modification, upstream and downstream of a point or nonpoint heat source, or before and after a modification that might impact stream temperature. Temperatures must be for similar time frames such as weeks or seasons. The larger the data set, the finer the precision in determining whether a material increase in stream temperature has occurred.

Currently the MPCA is evaluating mostly cold water fisheries for temperature-caused impairment because of the special sensitivity of cold water fish to elevations in temperature in streams.

e) Biological indicators

The presence of a healthy, diverse, and reproducing aquatic community is a good indication that a lake, stream, or wetland supports the aquatic life beneficial use. The aquatic community integrates the cumulative impacts of pollutants, habitat alteration, and hydrologic modification on a water body over time. Monitoring the aquatic community, or biological monitoring, is therefore a relatively direct way to assess aquatic life use-support. Interpreting aquatic community data is accomplished using an index of biological integrity or IBI. The IBI incorporates multiple attributes of the aquatic community, called "metrics," to evaluate a complex biological system. MPCA has developed fish and invertebrate IBIs to assess the aquatic life use of rivers and streams statewide in Minnesota as well as plant and invertebrate IBIs to assess depressional wetlands. A fish IBI has been developed by the DNR with assistance from MPCA to assess the aquatic life use of several lake types. A predictive model based plant indicator also developed by DNR as a measure of eutrophication stress to lake plant communities was used as supporting information only (Bacigalupi 2021).

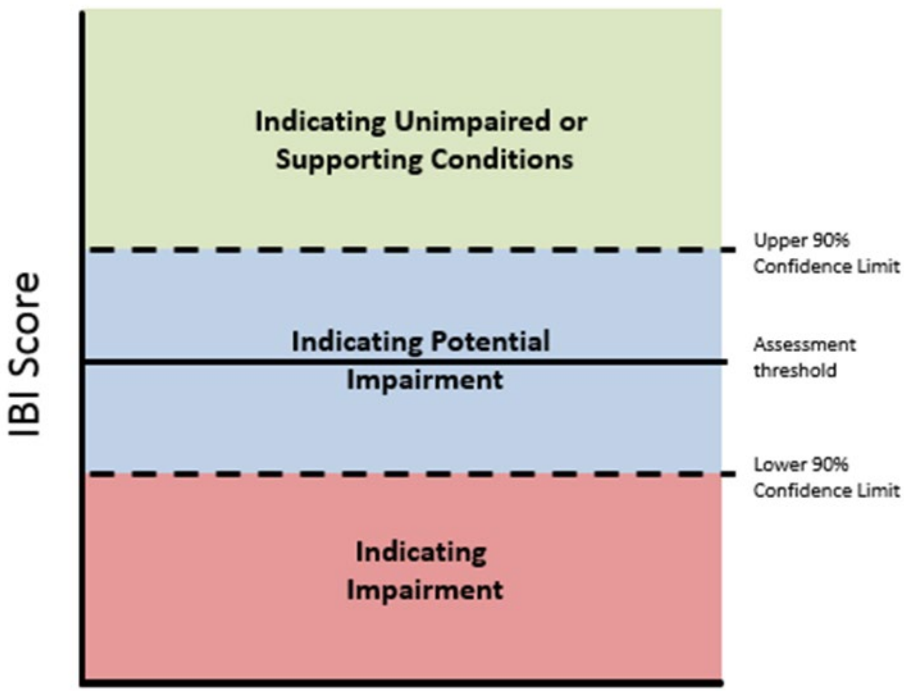
Further interpretation of aquatic community data is provided by an assessment threshold or biocriteria against which a stream IBI score can be compared. In general, an IBI score above this threshold is indicative of aquatic life use-support, while a score below the threshold is indicative of non-support.

Minnesota adopted Tiered Aquatic Life Uses (TALU) for streams and rivers in 2018. This framework refines Minnesota's single goal for aquatic life into three tiers, based on a water body's potential to support aquatic life. These tiered uses are Exceptional, General (current goal), and Modified. The process for determining the appropriate tier is called a Use Attainability Analysis (see http://www.pca.state.mn.us/index.php/view-document.html?gid=23281) and it is carried out before the assessment process. The mechanism for performing a biological assessment under the TALU framework is similar to the process for any other biological assessment, with the major difference being the biocriteria threshold (Table 12) used.

Bracketing each IBI assessment threshold is a 90% confidence interval that is based on the variability of IBI scores obtained at sites sampled multiple times in the same year (i.e., replicates). Confidence intervals account for variability due to natural temporal changes in the community as well as method error. For assessment purposes, sites with IBI scores within the 90% confidence interval are considered "potentially impaired." Upon further review of available supporting information, an IBI parameter review may change to "indicating support" or "indicating impairment" depending on the extent and nature of this additional information (Figure 2).

See Appendix F for further information regarding the basis of biological assessments including Minnesota's WQS, the development of the BCG, the selection of river and stream reference sites, and the development and the application of the IBI for lakes.

Figure 2. General diagram illustrating the characterization of individual biological indicator results.



f) River eutrophication

The River Eutrophication Standard (RES) is a two-part standard. An impairment listing requires an exceedance of the causative variable (total phosphorus) and a response variable that indicates the presence of eutrophication (i.e., undesirable levels of sestonic or suspended algae, benthic or attached algae, or excessive rooted vegetation). This response can be measured directly with chlorophyll-*a* (seston or periphyton) or indirectly via diel (daily) DO flux, five-day biochemical oxygen demand (BOD_5), or pH. These measures are highly correlated with each other in rivers and are indicators of stress for aquatic communities.

The first step in applying Minnesota's RES is to determine the appropriate River Nutrient Region (RNR), or regions, for the water body being assessed (see Appendix G). River eutrophication standards vary by RNR and can be found in Table 4**.**

Table 4. Minnesota's River eutrophication and standards by nutrient region. See Appendix G, Table 4 for map of regions.

	Causative	Response (stress)			
Region or river	**Total phosphorus µg/L**	**Chlorophyll-*a* (seston) µg/L**	**Diel dissolved oxygen flux mg/L**	**Biological oxygen demand mg/L**	**Periphyton chlorophyll-*a* mg/m²**
Northern River Nutrient Region	50	7	3.0	1.5	150
Central River Nutrient Region	100	18	3.5	2.0	150
Southern River Nutrient Region	150	40	5.0	3.5	150

For assessment purposes this means the cause indicator (total phosphorus) and response indicators (chl-*a*, BOD_5, diel DO flux, or pH) are used in combination and not independently. The eutrophication rule clearly states the requirement that cause and response indicators must both be exceeded to indicate a polluted condition.

Data minimums and summarizations

For total phosphorus (TP), chlorophyll-*a* (seston), and BOD_5, the following are required:

- A minimum of 12 measurements per parameter within the 10-year assessment period (minimum 2 years required).
- Data compared to the standard is a seasonal average – June to September data only.
- If multiple values exist for a parameter along a given reach for a single day, a daily average will be calculated prior to determining a seasonal average.

For DO flux

- A minimum of a 4-day deployment is required – June to September.
- A minimum of two deployments over separate years in the assessment window is required.
- It is preferred that the deployments coincide with summers when chemistry is collected and that the deployment is taken during mid-late summer.
- Multiple deployments will be summarized separately.

For periphyton chlorophyll-*a*

- A minimum of 2 years of data is required within the last 10 years.

For pH

- Class 2A waters: pH range is 6.5 ≤ concentration ≤ 8.5.
- Classes 2B and 2Bd waters: pH range is 6.5 ≤ concentration ≤ 9.0.
- Minimum of 20 samples necessary to indicate standard is met.
- Review of data is limited to June to September.

Assessment considerations

A stream is considered to exceed the river eutrophication standard if:

1. The TP concentration exceeds the standard; and
2. Chlorophyll-*a* (sestonic), BOD_5, DO Flux, pH OR periphyton exceeds the standard.

A stream is considered to meet the river eutrophication standard if:

1. The TP concentration meets the standard.
2. TP meets the standard and any available response variables meet the standard (this includes the situation where no response variables are present). Not all response variables must be available to consider the reach to be meeting the river eutrophication standard.
3. TP exceeds the standard and all response variables are available in sufficient quantities (chl-*a*, BOD_5, DO Flux, pH) and they all meet the standard.

A stream is considered to have insufficient information to determine if the river eutrophication standard is met if:

1. There are less than 12 samples of TP.
2. A sufficient TP data set exceeds the standard **and** no response variables meet the minimum data requirements.
3. The causative and/or response variables are within the standard error of the mean and confidence does not exist in determining whether the reach meets or exceeds the standard.
4. The causative and/or response variables have low data confidence or are not representative of ambient conditions (poor QA/QC, flood or drought biased sampling, proximity to continuously discharging facilities, etc.).

Due to the complexity of the standard, additional information to aid an assessment decision is available in Appendix G.

2. Data requirements and determination of impaired condition

Overall assessment of whether an AUID adequately supports aquatic life involves the review of the parameter-level evaluations and data quality in conjunction with all available supporting information (flow/water level, habitat, precipitation, plant surveys, etc.) to make an overall use-support determination. For a given AUID there may be chemistry indicator data, biological indicator data, or both types of data available for assessment. The final assessment takes into consideration the strength of the various indicators and the quality of the data sets and, in addition, looks at upstream and downstream conditions to gain a better understanding of the interactions between the individual AUID and the larger water body and watershed.

In general:

- A stream reach or lake is considered to be fully supporting of aquatic life if:
  - IBI scores for all available assemblages indicate fully supporting conditions, OR
  - The standards for river eutrophication and/or DO are met, and TSS/Secchi tube are met (streams only), AND
  - Other lines of evidence considered comprehensively, including upstream/downstream conditions, do not contradict a finding of full support.
- A stream reach or lake is considered to be **not supporting** if:
  - IBI scores for at least one biological assemblage indicate impairment, OR

- One or more water chemistry parameters indicates impairment, AND
- Other lines of evidence considered comprehensively, including upstream/downstream conditions, do not contradict a finding of non-support.
- If the above criteria are not met and the assessment is inconclusive, the result is a determination of insufficient **information** to indicate aquatic life use support.

In cases where an assessment unit has been determined to be not supporting based on biological indicators, water-chemistry parameters are added to the list of impairments only when the chemical impairment is clear enough that the AUID would be considered impaired even without the biological evidence.

The following paragraphs provide more details of the considerations that occur when analyzing the available data and information to make a comprehensive aquatic life use-support assessment, based on what types of indicator data are available. This information is used by the WAT and PJG for each watershed as guidance in making use-support decisions.

a) Only biological indicator data available

Fully Supporting – All **available** fish and invertebrate IBI scores within the assessment unit fall above the upper 90% confidence limit. A fully supporting determination does not require that both indicator assemblages have been measured within the assessment unit.

Not Supporting – All fish and/or invertebrate IBI scores fall below the lower 90% confidence limit. A not supporting determination does not require agreement between the indicator assemblages; one assemblage indicating impairment is sufficient for a not supporting determination.

Otherwise, the use assessment is potentially impaired when one or more IBI scores fall within the 90% confidence interval that bounds the assessment threshold **or** multiple IBI scores within an indicator assemblage are resulting in discrepant assessments. Further analysis is required to make a use-support determination, considering the following factors:

- Co-occurrence of indicator data.
- Habitat conditions.
- Sampling conditions.
- Watershed context.

b) Only water chemistry indicator data available

Fully Supporting (streams only) – 1) The standards for river eutrophication and/or DO are met, AND TSS/Secchi Tube are met for streams, AND 2) supporting information, including upstream/downstream conditions, does not strongly contradict a finding of full support. Making this determination requires considering the following factors:

- Co-occurrence of indicator data.
- Strength of indicator.
- Parameter-level evaluations.
- Sampling conditions.
- Watershed context.
- Continuous monitoring data (when available).

Not Supporting (streams or lakes) – 1) One or more water chemistry parameters indicate potential impairment or impairment **and** 2) supporting information including upstream/downstream or watershed conditions does not strongly contradict a finding of non-support. If the first condition is met, condition two should primarily be evaluated considering:

- Strength of indicator.
- Parameter-level evaluations.
- Watershed context.
- Continuous monitoring data (when available).

In general, information from within the assessment unit (strength of indicator and parameter-level evaluation) serves as the primary arbiter for making a not supporting determination, while assessments and data from adjacent assessment units (watershed context) provides additional information that either corroborates or refutes this determination. Considering these three factors together, a not supporting determination is more likely in situations where:

- Parameter-level evaluations indicate potential impairment or impairment,
- The indicators are strong, **and**
- The assessment is corroborated by similar conditions upstream or downstream of the assessment unit in question.

Continuous monitoring data, if available, can be used to either corroborate or refute the evidence provided by grab-sample data sets.

c) Both biological and water chemistry indicator data

Fully Supporting – 1) IBI score for at least one biological assemblage indicates supporting conditions OR the standards for river eutrophication and/or DO are met, and TSS/Secchi tube are met (streams only), **and** 2) other data and information considered comprehensively, including upstream/downstream or watershed conditions, do not strongly contradict a finding of full support. If the first condition is met, condition two should be evaluated considering the following factors:

- Co-occurrence of indicator data.
- Strength of indicator.
- Parameter-level evaluations.
- Habitat conditions.
- Sampling conditions.
- Watershed context.
- Continuous monitoring data (when available).

Not Supporting – 1) IBI score for at least one biological assemblage indicates impairment OR 2) IBI score for at least one biological assemblage indicates potential impairment **and** the parameter-level evaluations and other data and information considered comprehensively corroborate a finding of non-support OR 3) one or more water chemistry parameters indicate impairment **and** the evidence considered comprehensively leads to a conclusion of non-support. To evaluate all three conditions, consider the following factors:

- Co-occurrence of indicator data.
- Strength of indicator.
- Parameter-level evaluations.
- Habitat conditions.
- Sampling conditions.
- Watershed context.

- Continuous monitoring data (when available).

d) Insufficient information

If the criteria are not met for a fully supporting or not supporting assessment and the assessment is inconclusive, the result is a determination of insufficient information. “Insufficient information” determinations include situations where sufficient data are not available to assess the use, or the strength of the available indicator(s) is low and there is no supporting information available to help verify what the weak dataset is indicating. Sites receiving an “insufficient information” assessment may be prioritized for follow-up monitoring during MPCA stressor identification efforts, addressed by local monitoring efforts, or monitored further during the next round of IWM.

VI. Aquatic consumption and drinking water

This section focuses on Human Health-based Water Quality Standards (HH-WQSs): Class 2 chronic standards (CSs) and site-specific chronic criteria (CC). These standards serve as the basis for developing chronic or long-term protection for humans from toxic pollutants to ensure the beneficial uses of drinking water (where designated) and fish consumption and recreation in all surface waters are met. Class 2 numeric WQS and criteria for human health cover elemental and synthetic chemical contaminants characterized as toxic pollutants (as defined in Minn. Statute 115.01 Subd. 20).

The Class 1 designation and associated domestic consumption (DC) standards specifically address drinking water and food processing use of groundwater and designated surface waters. The Federal Safe Drinking Water Act standards incorporated by reference into Minn. R. 7050 provide the numeric basis for protecting this use. Application of Class 1 DC standards for nitrate and nitrite in surface waters designated for drinking water protection is also discussed in this section.

A. Pollutants with Class 2 human health-based chronic standards

Class 2 chronic standards (CS) to protect human health are developed for application in water, as described in the previous chapter, and in fish tissue. For toxic pollutants detected in surface water that lack CSs, the methods in Minn. R. 7050 and 7052 are used to develop site-specific CC. Fish tissue-based CSs (or CC) are described later in this section. Full details on HH-WQS are found in Minn. R. 7050 and 7052 and in the *Human Health-based Water Quality Standards Technical Support Document* (June 2017), available at https://www.pca.state.mn.us/sites/default/files/wq-s6-12a.pdf.

1. Algorithms for human health-based chronic standards

HH-WQS are set at concentrations to protect human users of surface waters. That protection considers the toxicity (deleterious, noxious, or injurious) characteristics of the pollutant and how much a population may be exposed to that pollutant through the three designated beneficial uses of surface waters: drinking water, recreational activities, and fish consumption. In short, HH-WQS encompass a pollutant's toxicity and a population's potential exposure and lead to numeric CSs (or site-specific CC) that cannot be exceeded in surface water or fish tissue.

The methods used to develop pollutant-specific numeric HH-WQS (Class 2 CSs or CC) for toxic pollutants were first adopted in 1990 for statewide application and in 1998 for the Lake Superior Basin. Currently, Minn. R. 7050 contains Class 2 standards for 69 toxic pollutants. Of these, 36 standards are more restrictive to protect human health than aquatic life (Minn. R. 7050.0222). Minn. R. 7052 contains Class 2 standards for
29 pollutants; for 15 of these standards human health is the basis for the most stringent CS (Minn. R. 7052.0100).

2. Pollutants with human health-based chronic standards

The pollutants that have human health-based CSs that are most often included in MPCA water quality assessments are briefly described. Pollutants other than those mentioned here may be assessed also, as data allow.

a) Trace metals

Trace metals with chronic standards to protect human health include antimony, arsenic, cobalt, mercury, and thallium. Mercury is discussed in the next section. Minn. R. 7050 and 7052 provide human health-based CSs for trace metals. To determine if human health-based CSs are being met, data with the total sample fraction is used. Both dissolved and total metals measurements can be used to determine impairment, but dissolved metals data cannot be used to determine if standards are met. See also Figure 1.

b) Mercury

Mercury is the classic example of a bioaccumulative element; it never degrades and it can bioaccumulate through the food chain to reach toxic levels in many fish species, which if eaten in high amounts, can lead to serious health effects. Neurodevelopmental effects to children exposed during gestation are of most concern. WQS alone cannot reduce mercury to levels that are not of concern for fish consumers, so the Minnesota Department of Health (MDH) provides Fish Consumption Guidance as an important means for providing information to fish consumers to keep exposure from mercury and other bioaccumulative pollutants in fish low (discussed further in the fish pollutant section).

Mercury CSs are based on total concentrations and, thus, total mercury measurements are used in assessments. The Clean Hands/Dirty Hands sample collection technique is required for low-level mercury analysis and determination of the CSs (EPA. 1996. Method 1669). Only mercury data collected by this technique are used for assessments. Minnesota has two water-column Class 2 WQS for total mercury, as shown below (although the more stringent CS for Lake Superior is based on fish-eating wildlife, this value is protective of human consumers and assessed the same way as the statewide mercury CS).

- 6.9 ng/L chronic standard, Minn. R. 7050.0222 (statewide)
- 1.3 ng/L chronic standard, Minn. R. 7052.0100 (waters of Lake Superior Basin)

Minnesota also has a fish tissue mercury standard in Minn. R. 7050 discussed in Section VI. B. 2.

c) Pesticides

The MDA conducts extensive pesticide monitoring in surface waters and submits all data to the MPCA for assessments. At present, the MPCA has human health-based CSs for alachlor, atrazine (including degradates), 2,4-D, 2,4,5-TP, carbofuran, glyphosate, methoxychlor, picloram and simazine.

3. Data requirements and determination of impaired condition

The requirements for assessing water bodies for exceedances of human health-based CSs are essentially the same as for chemicals with aquatic life toxicity-based CSs (see Section V. A.) The major difference is that data compared to the human health-based CSs are averaged over a 30-day period.

Table 5. Summary of data requirements and exceedance thresholds for assessment of pollutants with human health-based, human health- and aquatic life toxicity-based, and wildlife-based standards.

Period of record	Use-support or listing category	
Most recent 10 years	No more than 1 exceedance of the Chronic Standard in 3 years, and no exceedances of the Maximum Standard: Not listed	2 or more exceedances of the Chronic Standard in 3 years, or 1 or more exceedances of the Maximum Standard: Listed

B. Protection for human consumption of fish

This section describes the assessment of fish for human consumption based on fish contaminant data. The MPCA has methods to develop fish tissue CSs (or CC) to use as the basis of impairment decisions - determining if pollutants in fish fillets exceed HH-WQS. The prior practices of applying MDH Fish Consumption Advice or Guidance thresholds is limited as described in this section. Most fish monitoring data is collected through the interagency Fish Contaminant Monitoring Program (FCMP), which also provides the data used by MDH for Fish Consumption Guidance and Safe-Eating Guidelines. See Minn. R. 7050.0219, https://www.revisor.mn.gov/rules/?id=7050.0219, for details.

1. Basis for assessment of fish contaminants

The basis for assessing the contaminants in fish tissue is the narrative WQS and assessment factors in Minn. R. 7050.0150, subp. 7, which states the following:

Subp. 7. Impairment of waters relating to fish for human consumption.

A. In evaluating whether the narrative standards in subpart 3, which prevent harmful pesticide or other toxic pollutant residues in aquatic flora or fauna, are being met, the commissioner must use the methods in:

(1) parts 7050.0218 and 7050.0219 for site-specific fish tissue-based chronic criterion (CC_{ft}); or
(2) parts 7050.0222 and 7052.0100 for fish tissue-based chronic standard (CS_{ft}).

B. If CS_{ft} has not been established for a pollutant with chronic standards (CS) applicable in water (CS_{dfr}, CS_{dev}, or CS_{fr}, as defined in parts 7050.0218, subpart 3, item Q, and 7050.0219, subpart 13, item B), the residue levels in fish muscle tissue established by the Minnesota Department of Health must be used to identify surface waters supporting fish for which the Minnesota Department of Health recommends a reduced frequency of fish consumption for the protection of public health. A water body will be considered impaired when the recommended consumption frequency is less than one meal per week, such as one meal per month, for any member of the population. That is, a water body will not be considered impaired if the recommended consumption frequency is one meal per week, or any less restrictive recommendation such as two meals per week, for all members of the population. The impaired condition must be supported with measured data on the contaminant levels in the resident fish.

C. When making impairment determinations in an individual water body for a pollutant with both a fish tissue-based CC_{ft} or CS_{ft} and a CS applicable in water, comparison of fish tissue data to the CC_{ft} or CS_{ft} must be the basis for the final impairment determination.

2. MPCA assessment of pollutants in fish tissue

Chemicals that persist in the environment and "build up" in the tissues of aquatic organisms to higher concentrations than the concentrations in the surrounding water are called bioaccumulative chemicals of concern (BCCs). Uptake through the food chain means that at each step, from plants to prey to predator, the concentrations in the biota increase. This "biomagnification" is a concern because many game fish (e.g., walleye, northern pike, bass, and lake trout) are at the top of the aquatic food chain and typically carry the highest tissue concentrations of a BCC in the aquatic system. The pollutants then pass to the humans that consume these fish.

The bioaccumulation factor (BAF) is the ratio between the concentration of the chemical in the biota and the concentration of the chemical in the water. BAFs can exceed one million (meaning the concentration in the biota is more than one million times higher than the concentration in the water) for very highly bioaccumulative chemicals. A BAF must be determined in order to calculate a human health-based water column standard. For pollutants defined as BCCs, or those with BAFs > 1000, the resulting CSs are very low water column concentrations. These low water column concentrations of pollutants are needed in order to limit the pollutant concentration in fish tissue. For these chemicals, such as mercury, polychlorinated biphenyls (PCBs), and dioxins, human exposure from the fish consumption pathway also far exceeds that from drinking water or recreational activities.

The MDH has a long-standing public outreach program to support the health of people that eat fish in Minnesota. The MDH issues safe-eating guidelines for how often certain fish can be eaten to gain the most health benefits from this important source of protein, while minimizing exposure to toxic pollutants (BCCs). The guidelines address mercury, PCBs, and perfluorooctane sulfonate (PFOS) using concentrations in fish tissue that corresponds to meal frequency recommendations (Table 6). The advice also identifies water bodies with fish showing elevated levels of dioxins and furans. The goal of the Safe-eating Guidelines is to help people make informed decisions on which fish to eat and which to avoid. The advice is not mandatory nor regulatory.

Table 6. Fish tissue concentrations for levels of consumption advice established by MDH (2017 to present)

Consumption Advice[1]	Four Meals/Week[2]	One Meal per Week	One Meal per month	One Meal per Two Months	Do Not Eat
Mercury (mg/kg)	≤ 0.05	>0.05 - 0.22	>0.22 - 0.95		> 0.95
Total PCBs (mg/kg)	≤ 0.05	>0.05 - 0.22	>0.22 - 0.95	>0.95 - 1.89	> 1.89
PFOS (mg/kg)	≤ 0.01	>0.01 – 0.05	>0.05 – 0.20		> 0.20

[1]Consumption advice for young children and women who are pregnant or may become pregnant: https://www.health.state.mn.us/communities/environment/fish/.

[2]As of May 2021, "MDH Statewide Safe-Eating Guidelines for the General Population have changed from unrestricted to four servings per week for the panfish group of fish species. This change was made to clarify what is meant by "unrestricted" and to take into account findings of low levels of Perfluorooctane Sulfonate (PFOS) in fish throughout Minnesota."

The MPCA has adopted methods to develop fish-tissue standards or site-specific criteria for BCCs identified in or with the potential to be present in fish. When developed, the site-specific criteria (CC_{ft}) are used in place of the MDH thresholds. Details on site-specific CC_{ft} and applicable water bodies are found at https://www.pca.state.mn.us/water/site-specific-water-quality-criteria.

a) Mercury

Mercury is a BCC detected in most fish. Concentrations reach levels of concern in many predator species. Based on EPA guidance, MPCA adopted a fish tissue standard for mercury in 2008 to provide a more accurate and directly usable standard to protect fish consumers.

The fish tissue-based CS_{ft} for total mercury is found in Minn. R. 7050.0222. It is applicable in all Class 2 surface waters.

- 0.2 mg/kg in edible fish tissue (statewide)

b) Polychlorinated biphenyls (PCBs)

PCBs constitute a group of chlorinated organic compounds distributed worldwide. Their extensive historical use combined with their persistence, bioaccumulative properties, and cancer and non-cancer toxicity, make them very serious environmental pollutants. Concentrations of PCBs in water are very low (typically less than one part per trillion) and difficult to measure. However, because they bioaccumulate as much as a million fold or more in fish, they are readily measured in fish tissues. PCBs are usually assessed for the 303(d) List on the basis of their presence in fish.

Previous and ongoing assessments of PCBs in fish tissue will use the fish consumption guidance concentration threshold that restricts fish consumption from one meal a week to one meal a month, 0.22 mg/kg, for determining if the fish meet the protection level goals for fish consumers. Concentrations above this amount identified as advisory levels for any fish species in a water body result in that water body being listed as impaired for PCBs.

While water CSs exist in rule, CS_{ft} in fish tissue need to be adopted, along with revised CSs in water, in a future rulemaking, before they can be used for assessment.

c) Perfluorooctane sulfonate (PFOS)

PFOS is a synthetic perfluorinated chemical used for decades to make products that resist heat, oil, stains, grease, and water. The MPCA has been monitoring for PFOS in fish since 2004.

Certain waters are identified as impaired due to average concentrations in fillets exceeding the threshold used by MDH to issue guidance restricting fish consumption to one meal a month. Waters listed as impaired prior to 2017 were based on a threshold of 0.20 mg/kg, set by MDH in 2009. Waters listed as impaired after that time are based on a threshold of 0.05 mg/kg (50 ng/g or 50 ppb) as set by MDH in 2017.

With the adoption of the revised methods for HH-WQS, MPCA has developed site-specific criteria (CC_{ft}) for PFOS, and will continue to move away from using MDH fish consumption guidelines for assessment. The criterion is 0.37 ng/g PFOS in fish tissue. Assessments for waters where the site-specific CC_{ft} apply are based on this PFOS criterion developed using the methods in Minn. R. 7050.0217 to 7050.0219. Details on site-specific CC_{ft} and applicable water bodies are found at https://www.pca.state.mn.us/water/site-specific-water-quality-criteria.

d) Dioxins and Furans

Dioxins and furans are similar to PCBs in many respects. Both represent a family of chlorinated organic chemicals, some of which are very persistent, bioaccumulative and toxic. They are global in their distribution. Unlike PCBs, dioxins and furans were never intentionally manufactured. The major sources are combustion of waste, plastics, and wood, chlorine bleaching of pulpwood (now largely phased out), and trace contaminants in other manufactured organic compounds. 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) has been shown to be carcinogenic in animals at extremely low doses. The MPCA has Class 2 HH-WQS for 2,3,7,8-TCDD in Minn. R. 7052, applicable only to waters in the Lake Superior basin. These standards also include other dioxins and furans with toxic equivalent factors. Some PCB congeners can also have dioxin-like toxicity and considered when data are available. The only 2,3,7,8-TCDD standard in Minn. R. 7050 is the EPA drinking water standard of 30 pg/L.

The MPCA evaluates waters for dioxins and furans only at site-specific locations where contamination is suspected or where data are needed to support remedial efforts. Evaluation of dioxin and furans in fish tissue will be based on site-specific CC or CSs developed based on Minn. R. 7050.0217 to 7050.0219 and Minn. R. 7052.0270.

3. Data requirements and determination of impaired condition

The 303(d) Impaired Waters List identifies water bodies that do not meet legally enforceable water quality standards (WQS) or site-specific criteria, and for which a remedial plan may be required. An important caveat is that one cannot assume, because a particular water body does not appear on the 303(d) List, the fish in that water body are safe for unlimited consumption. Most likely, it means the fish from that water body have not been tested. Only those water bodies from which the fish have been tested and found to exceed the impairment thresholds are put on the 303(d) List. In addition, water bodies listed as impaired for fish consumption can still yield fish low in pollutant concentrations. The MDH safe-eating guidelines should be consulted for advice on fish consumption on a statewide or water body basis (MDH 2021).

The MPCA applies the MDH guidance threshold concentrations summarized in Table 6 to the most recent 10 years of data from a water body. Impairments for PCBs are based on a fish tissue concentration exceeding 0.22 mg/kg and 0.05 mg/kg for PFOS in water bodies without CC_{ft}; these are the upper thresholds for one meal per week fish consumption.

For pollutant data in fish that rely on CS_{ft} or CC_{ft}, the determination of impaired waters for fish consumption reflects approaches used to assess water quality data. The 0.2 mg/kg fish mercury concentration is the threshold for determining impairment for total mercury in edible fish tissue. Average fish tissue concentrations that exceed 0.20 mg/kg and are equal to or less than 0.572 mg/kg fall into the range for the EPA-approved statewide mercury TMDL. Waters with concentrations greater than 0.572 mg/kg are added to the TMDL List. For other fish pollutants, the MPCA may develop site-specific CC_{ft} or future CS_{ft} to assess fish for impairment.

A water body is defined as impaired based on one of the two following approaches depending on the number of fish and species with available monitoring data.

a) Multiple fish of one species:

If more than 10% of the fish (minimum of five fish) in a species are greater than the fish tissue-based CS_{ft} or CC_{ft}, the fish are not meeting the WQS. This is equivalent to saying the water is impaired if the 90th percentile of the pollutant concentration for any fish species is greater than the CS_{ft} or CC_{ft} . This is the same protocol that has been used to assess mercury in fish.

To determine which water bodies (lake, reservoir, or stream assessment unit) are impaired for fish consumption, the Minnesota FCMP database is queried for the following criteria:

- Fish collected in the last 10 years, unless the 90th percentile between years 10 to 6 and years 5 to present are statistically different, in which case only the most recent 5 years is used in the assessment.
- Filet with or without skin on; no whole fish.
- At least five fish in a species, including fish within a composite sample, is needed for 90^{th} percentile calculation.
- 90th percentile fish tissue concentration is greater than CS_{ft} or CC_{ft} (i.e., more than 10% are greater than CS_{ft} or CC_{ft}).

The 90th percentile rank is calculated by multiplying the number of fish by 0.9 and rounding to the nearest whole number. The 90th percentile pollutant concentration is determined for each water body-species by (1) ranking the samples within each water body-species from low to high, (2) concentration of a composite sample is treated as the concentration for all fish within the composite, (3) if the 90th percentile ranked fish is greater than CS_{ft} or CC_{ft} or is in a composite that is greater than CS_{ft} or CC_{ft}, it is marked as impaired.

b) Fewer fish of more than one species:

If a water body has multiple species of fish with pollutant monitoring data, but fewer than five fish per species, the alternate method for determining if WQS are being met is through averaging a concentration across species. In a weight-of-evidence approach, if the average concentration of at least three species exceeds the CS_{ft} or CC_{ft} that water body would also be identified as impaired. Like the evaluation for multiple fish of one species, fish collected in the last 10 years would be used unless enough fish samples are available to compare average concentrations between years 10 and 6 and 5 to present. If the averages are statistically different then only the most recent 5 years is used in the assessment.

Both scenarios recognize that concentrations in fish are a result of a longer-term average exposure and that the fish sampled by the FCMP focus on those species regularly caught and consumed by Minnesotans; reasonable evidence of fish with pollutant concentrations above CS_{ft} or CC_{ft} warrants concern and impairment designation. Based on the FCMP sampling protocol, most water bodies monitored will exceed the minimum data requirements or include the species of most concern for the respective pollutant (i.e., walleye for mercury or bottom feeders, such as carp or catfish, for PCBs).

With the revised HH-WQS methods, fish data for BCCs is the basis for impairment determination if water data are also available (Minn. R. 7050.0222, subp. 7, item C.).

C. Additional guidance for assessing human health-based standards

1. Chemical breakdown products or environmental degradates

Some pollutants, when introduced into the environment, undergo chemical transformation through microbial, photolysis, or other processes. Particularly for pesticides, there are known common environmental breakdown products referred to as degradates that originate from the “parent” chemical. In order to be health protective, breakdown chemicals that originate from a “parent” chemical are assessed the same as the “parent” when toxicological data on the degradate are insufficient for a chemical-specific health based water value. To address degradates found in surface water, the MPCA applies the parent HH-WQS to environmental degradates or MDH health-based guidance when available (Minn. R. 7050.0222, subp. 7, item D).

2. Mixtures of pollutants in a water or fish sample

Another aspect to assessing Class 2 CSs (and CC) based on human health is the presence of more than one toxic pollutant in a sample. This is dependent on the toxicity determination of each pollutant: carcinogen, denoted with a “(c)” next to the pollutant’s name in Minn. R. 7050.0220 or 7050.0222, or noncarcinogen.

For linear carcinogens, the additivity algorithm is as listed in Minn. R. 7050.0222, subp. 7 item E, and Minn. R. 7052.0230, subp. 2. The additivity equation applies to chemicals that are linear carcinogens and have HH-WQS calculated with a cancer slope factor. A risk index is calculated for each carcinogen in the sample by dividing the concentration of the pollutant by its CS (or CC) and summing those values. The risk index value has to be equal to or less than one to meet HH-WQS. An index that exceeds one indicates the excess cancer risk level is greater than 1 in 100,000 and is in violation of the HH-WQS.

The MPCA recently added to this existing protection to surface water users by including a new approach for noncancer mixtures: an additivity analysis modeled on the MDH Health Risk Limit rule. The approach is again based on summing up the ratio of each pollutant concentration measured in the surface water or in fish tissue to their respective CS (or CC) based on their *Health Endpoint*. To ensure total exposure does not exceed the threshold for noncancer effects in the target organ, system, or process (development), the sum or *Health Risk Index* has to equal one or less to meet the HH-WQS.

Health Risk Index Endpoints (*Health Endpoints*) will be incorporated into HH-WQS for evaluation of mixtures of noncarcinogens. The MDH lists *Health Endpoints* for each noncarcinogen (or nonlinear carcinogen) unless the available study used to develop the toxicological values (reference dose) did not identify a specific adverse effect. *Health Endpoints* identify the most sensitive target organs or systems (e.g., nervous) or developmental process affected by that pollutant. These endpoints are used to group chemicals to evaluate mixtures if more than one pollutant with the same adverse effect is measured in a fish sample or water body. The details of this evaluation are in Minn. R. 7050.0222, subp. 7, item D.

D. Class 1 drinking water standards for nitrate nitrogen

Class 1 waters are protected as a source of drinking water (Minn. Rule 7050.0221). In Minnesota, all groundwater and selected surface waters are designated Class 1. The assessment of groundwater (Class 1A), where treatment is not necessary to meet federal drinking water standards, is outside the scope of this Guidance. The MDH monitors municipal finished water supplies for compliance with drinking water standards. The assessment of Class 1B and 1C listed surface waters for potential impairment by nitrate nitrogen is discussed in this section.

1. Nitrate nitrogen

Nitrate nitrogen poses a risk to human health at concentrations exceeding 10 mg/L in drinking water. Humans, especially infants under six months of age, who are exposed to nitrate in drinking water at concentrations exceeding the 10 mg/L federal safe drinking water standard (which is incorporated by reference into Minn. R. 7050.0221), can develop methemoglobinemia, a blood disorder that interferes with the ability of blood to carry oxygen.

The 10 mg/L standard is an acute toxicity standard. Long term, chronic exposure to nitrate in drinking water is less well understood but has been linked to the development of cancer, thyroid disease, and diabetes in humans.

In recognition of the trend of increasing nitrate concentrations in Minnesota streams and the public health and economic impact arising from elevated nitrate concentrations in drinking water (a particular concern in Southeast Minnesota's karst region), the MPCA assesses Class 1B and 1C designated surface waters for potential impairment by nitrate nitrogen.

2. Data requirements and determination of impaired condition

When assessing drinking water-protected surface waters designated as Class 1B and 1C, MPCA compares 24-hour average nitrate concentrations to the 10 mg/L standard. Two 24-hour averages exceeding 10 mg/L within a three-year period indicates impairment.

Single measurements of nitrate concentrations under relatively stable conditions are generally considered to be sufficiently representative of 24-hour average concentrations for the purpose of assessments. When concentrations are more variable, multiple samples or time-weighted composite samples may be necessary in order to calculate a sufficiently accurate average concentration. The necessary number and type of samples can vary considerably from one situation to another and the determination of adequacy for the purpose of assessment will necessarily involve considerable professional judgment.

Table 7. Summary of data requirements and exceedance thresholds for assessment of nitrate nitrogen, Class 1 drinking water standard.

Period of record	Use-support or listing category	
Most recent 10 years	No more than 1 exceedance of the acute standard in 3 years: Not listed	2 or more exceedances of the acute standard in 3 years: Listed

VII. Protection of wildlife in the Lake Superior Basin

Protection of the aquatic life use includes the protection of wildlife consumers of aquatic organisms. Minnesota has developed four wildlife-based WQS – all in Minn. R. 7052, the Great Lakes Water Quality Initiative (GLI) rule. The GLI rule focuses on the reduction of bioaccumulative toxic chemicals in the Great Lakes ecosystem as a whole. The standards in Minn. R. 7052 are applicable only to the surface waters of the Lake Superior basin in Minnesota. The GLI chronic wildlife-based standards are listed below:

- DDT – 11 pg/L.
- Mercury – 1300 pg/L.
- PCBs – 122 pg/L (GLI human health-based standards for PCBs are more stringent than the wildlife based standard).
- 2,3,7,8-TCDD – 0.0031 pg/L (GLI human health-based standards for dioxin are more stringent than the wildlife based standard for Lake Superior and Class 2A waters, but not for Class 2Bd and 2B, and 2D waters).

The assessment of water bodies for compliance with the GLI wildlife-based standards follows the same protocols used to assess water bodies for human health-based standards, as described in the previous section (Table 5).

VIII. Protection of aquatic recreation

This section addresses the assessment of water quality for pollutants that have aquatic recreation-based standards. Standards based on protecting the ability to recreate on and in Minnesota's waters are Class 2 standards. An overview of these standards and their application for assessment is provided below.

A. Streams and rivers – *E. coli* bacteria

The numeric standards in Minn. R. 7050 that directly protect for primary (swimming and other recreation where immersion and inadvertently ingesting water is likely) and secondary (boating and wading where the likelihood of ingesting water is much smaller) body contact are the *E. coli* (*Escherichia coli*) standards shown in Table 8. *E. coli* standards are applicable only during the warm months since there is very little swimming in Minnesota in the non-summer months. Exceedances of the *E. coli* standard mean the recreational use is not being met.

The MPCA uses an *E. coli* standard based on a geometric mean EPA criterion of 126 *E. coli* colony forming units (cfu) per 100 mL. *E. coli* has been determined by EPA to be the preferred indicator of the potential presence of waterborne pathogens.

Table 8. *E. coli* water quality standards for Class 2 and Class 7 waters.

Use class	Standard Number of organisms per 100 mL of Water		Applicable season	Use
	Monthly geometric mean[1]	**10% of samples maximum[2]**		**Body contact**
2A, trout streams and lakes, 2Bd, 2B, non-trout (warm) waters	126	1260	April 1 – October 31	Primary
2D, wetlands	126	1260	April 1 – October 31	Primary, if the use is suitable
7, limited resource value waters	630	1260	May 1 – October 31	Secondary

[1]Not to be exceeded as the geometric mean of not less than five samples in a calendar month.
[2]Not to be exceeded by 10% of all samples taken in a calendar month, individually.

1. Data requirements and determination of impaired condition

There is a considerable amount of *E. coli* data available in Minnesota, and also older fecal coliform data. For assessment purposes, only results analyzed within 24 hours of sample collection are used and only *E. coli* measurements are used. Data over the full 10-year period are aggregated by individual month (e.g., all April values for all 10 years, all May values, etc.). At least five values for each month is ideal, while a minimum of five values per month for at least three months, preferably between June and September, is necessary to make a determination. Assessment with less than these minimums may be made on a case-by-case basis.

Where multiple bacteria/pathogen samples have been taken on the same day on an assessment unit, then the geometric mean of all the measurements on that day will be used for the assessment analysis.

If the geometric mean of the aggregated monthly values for one or more months exceeds 126 organisms per 100 mL, that reach is considered to be impaired. Also, a water body is considered impaired if more than 10% of individual values over the 10-year period (independent of month) exceed 1260 organisms per 100 mL This assessment methodology more closely approximates the five-samples-per-month requirement of the standard while recognizing typical sampling frequencies, which usually only provide one sample in a single month, rarely five. Table 9 summarizes the assessment process.

Table 9. Assessment of water bodies for impairment of swimming use - data requirements and exceedance thresholds for *E. coli* bacteria.

Period of record	Minimum no. of data points	Use-support or listing category based on exceedances of the *E. coli* standard	
Standard exceedance thresholds → Monthly geometric mean > 126 orgs/100 mL (Class 2) > 630 orgs/100 mL (Class 7)		**No months**	**1 or more months**
Most recent 10 years	see text	Not listed	Listed
Standard exceedance thresholds → Exceeds 1260 orgs/100 mL*		**≤ 10 %**	**>10 %**
Most recent 10 years	15	Not listed	Listed

* In full data set over 10 years.

Expert review of the data provides a further evaluation. When fewer than five values are available for most or all months, the individual data are reviewed. Considerations in making the impairment determinations include the following:

- Dates of sample collection (years and months).
- Variability of data within a month.
- Magnitude of exceedances.
- Remark or data qualifier codes associated with individual values.
- Previous assessments and 303(d) listings.

In some circumstances where four values are available for some or all months, a mathematical analysis is done to determine the potential for a monthly geometric mean to exceed the 126 organisms/100mL standard. All assessments are reviewed by the Watershed Assessment Team (WAT) for each watershed.

Large datasets

Aggregating data by month across years for very large datasets diminishes the value of the data and assessment, making it less likely that periodic *E. coli* exceedances will be identified that indicate impairment. Data aggregation should be held to a minimum, no more than necessary to have sufficient data to satisfy the requirements for determining exceedances.

Alternative methods of data analysis may be used based on a professional judgment review of the data. Where there are five values per individual month or 30-day time period, the data will not be aggregated and individual monthly or 30-day geometric means may be calculated. Alternatively, data may be aggregated by month across consecutive two-year or five-year time periods. If more than 10% of the geometric means calculated exceed the 126 org/100 mL standard, the AUID is assessed as not supporting aquatic recreation.

B. Great Lakes Shoreline (Lake Superior) beaches – *E. coli* bacteria

The Clean Water Act defines Coastal Recreation Waters as the Great Lakes and marine coastal waters (including coastal estuaries) that are designated under section 303(c) of the Clean Water Act for use for swimming, bathing, surfing, or similar water contact activities. The MPCA applies the coastal waters definition and Beaches Environmental Assessment and Coastal Health (BEACH) Act water quality standards to all bacteria monitoring sites on the Lake Superior shoreline and in the mouths of tributaries that are representative of shoreline/Lake Superior conditions. The St. Louis River and Duluth-Superior Harbor sites monitored in the BEACH Act program that extends upstream in the St. Louis River to the Boy Scout Landing Beach are also considered within the coastal recreation designation.

Lake Superior coastal waters are subject to *E. coli* WQS in the BEACH Act rule [November 2004 *Water Quality Standards for Coastal and Great Lakes Recreation Waters* rule (69 FR 67217, November 16, 2004), found at http://www.gpo.gov/fdsys/pkg/FR-2004-11-16/html/04-25303.htm]. These standards as applied in Minnesota are shown in Table 10.

Table 10. E. coli water quality standards for coastal recreation waters.

Standard No. of Organisms Per 100 mL of Water		Applicable Season	Use
Monthly Geometric Mean[1]	**10 % of Samples Maximum[2]**		**Body Contact**
126	235	April 1 – October 31	Primary

[1] Not to be exceeded as the geometric mean of not less than five samples in a calendar month.
[2] Not to be exceeded by 10% of all samples taken in a calendar month, individually.

1. Data requirements and determination of impaired condition

There is a considerable amount of *E. coli* data collected as part of the BEACH monitoring program in Minnesota. Most beaches are monitored weekly from Memorial Day to Labor Day, while some are monitored twice weekly. To ensure use of the most recent data, data for the most recent five year period are used and assessments are made every other (odd numbered) year.

When there are five or more samples per individual month or 30 day time period, individual monthly geometric means are calculated and compared to the 126 orgs/100mL standard for the period April 1 through October 31. If more than 10% of the geometric means calculated exceed the 126 orgs/100mL standard, or if more than 10% of the individual sample results in the entire dataset exceed the maximum criterion of 235 orgs/100mL, the AUID is assessed as not supporting aquatic recreation.

When sampling frequency results in smaller data sets, data is aggregated by month across years. If one or more of the monthly aggregated geometric means exceeds
126 orgs/100mL, or more than 10% of the individual sample results in the entire dataset exceeding the maximum criterion of 235 orgs/100mL, the AUID is assessed as not supporting aquatic recreation.

Data from adjacent sampling sites on the same beach are combined. For sites with both tributary mouth stations and BEACH stations, data from each station are assessed separately and the results considered using best professional judgment to make an assessment decision. For sites with only tributary mouth samples, the data are assessed against the coastal recreation water standards. Streams tributary to Lake Superior with bacteria data at stations upstream of the mouth are assessed as stream AUIDs using the statewide WQS and methodology in Part A. above.

The overall use-support assessment also requires best professional judgment to consider and integrate information regarding the timing, frequency, magnitude, and duration of exceedances along with other conditions present at the time of sampling. These longer-term use-support assessments based on several years of data are distinguished from the short-term beach advisory postings (water contact not recommended) that are based only on current 'real-time' data.

C. Lake eutrophication

Excessive nutrient loads, in particular total phosphorus (TP), lead to increased algae blooms and reduced transparency – both of which may significantly impair or prohibit the use of lakes for aquatic recreation. The ecoregion-based eutrophication standards are the primary basis for aquatic recreational use assessments in lakes.

1. Water body classification and ecoregion determination

As the eutrophication standards are specific to ecoregion and lake depth, a number of steps are required to be completed prior to the actual assessment of the water body. MPCA rules define lake, shallow lake, reservoir, and wetland (Minn. R. 7050.0150). The determination between the four requires an analysis of basin depth and littoral area. Additionally, a series of questions was developed to help make the differentiation between shallow lake and wetland. These can be found in Appendix D. This step of determining the appropriate standard includes a desktop review using GIS and available morphometric data and may include a site visit, if the decisions cannot be made from this review. Decisions are recorded and stored in the assessment database for future reference.

Reservoirs with residence times less than 14 days are not assessed as lakes, per EPA guidance (EPA 200a, Kennedy 2001). For this purpose, residence times are usually determined under conditions of low flow. A mean flow for the four-month summer season (June – September) with a once in 10-year recurrence interval is normally used. The MPCA may establish a minimum residence time of less than 14 days on a site-specific basis if credible scientific evidence shows that a shorter residence time is appropriate for that reservoir.

The majority of the lakes in the state (98%) reside in four of the seven ecoregions (EPA Omernik Level III ecoregions). The remaining 2% of lakes reside in one of three ecoregions: Red River Valley, Northern Minnesota Wetlands, and the Driftless Area (Heiskary and Wilson 2005). Percent land use by categories (forest, pasture/open, cultivated, urban, water/wetland) are calculated for the lake watershed using the most recent national land cover dataset. These percentages are then compared to the breakdown of land use for the standards development dataset to see which ecoregion is more similar to the lake in question. The next step involves comparing morphometry of the lake basin (large, small, deep, shallow); different ecoregions have different lake characteristics. This data is used together to determine the proper ecoregion-based standard to address these lakes that do not fall in the ecoregions for which criteria have been developed and for lakes that are near an ecoregion boundary. See Table 11 for Minnesota's ecoregion-based WQS.

2. Data requirements and determination of use assessment

a) Minimum data requirements

Samples must be collected over a minimum of two years and data used for assessments must be collected from June to September. Typically, a minimum of eight individual data points for TP, chlorophyll-*a* (corrected for pheophytin or corrected chl-*a*), and Secchi are required.

b) Lake assessment determinations

Data used for phosphorus and chlorophyll-*a* calculations are limited to those collected on the same day, from the upper most three meters of the water column (surface). If more than one sample is collected in a lake per day, these values are averaged to yield a daily average value. Following this step, all June to September data for the 10-year assessment window are averaged to determine summer-mean values for TP, corrected chl-*a*, and Secchi depth. These values are then compared to the standards and the assessment is made (Table 11).

Lakes where TP and at least one of the response variables (corrected chl-*a* or Secchi) exceed the standards are considered impaired. For lakes with excellent data quality (2+ years of data) and where all parameters are better than the standards, an assessment of full support is made. Lakes with good quality data (1-year data plus Secchi trends) may be considered for full support assessment as well. In this case, the assessment thresholds have been adjusted by 20% (made more stringent) and lakes with good quality data that meet these thresholds will be considered fully supporting. This modification of the thresholds provides a margin of safety to assure that lakes with lesser amounts of data are supporting the beneficial use.

In some instances, a lake may have good or excellent quality data but only one of the thresholds is exceeded (e.g., only TP or only corrected chl-*a* or Secchi). In this instance, the lake will be considered to have insufficient data to assess because both the cause (TP) and at least one response (chl-*a* or Secchi) must either meet to indicate support or both exceed to indicate impairment. For lakes that do not meet minimum data requirements and use-support cannot be determined, a determination of insufficient data will be made.

c) Reservoirs and other special situations

Sampling design and assessments for aquatic recreational use for reservoirs may be different from those used for lakes. Since reservoirs typically exhibit distinct zones, often referred to as inflow segment, transitional segment, and near-dam segment, calculation of "whole reservoir" mean TP may not be an appropriate basis for assessing aquatic recreational use. Rather, the MPCA may evaluate the status of the reservoir based on a specific segment – most likely the near-dam segment. In addition, water residence time may vary substantially as a function of river flow (e.g., Lake Pepin; Heiskary and Walker 1995) and may influence algal response to available nutrients. In addition, reservoirs often have very large watersheds that may drain portions of one or more ecoregion. Hence ecoregion-based standards based on where the reservoir is located may not always be the best basis for evaluating use-support.

Lakes with distinct bays, such as Lake Minnetonka, may present a similar situation. The bays (basins) may need to be assessed on an individual basis (data is stored by specific basin, not by whole lake). In some instances, a single bay may exceed the listing thresholds while other bays in the lake do not. In this case it should be determined whether the entire lake should be listed (e.g., there is distinct interaction between the bays) or simply the individual bay. This will likely require knowledge of flow-through patterns in the lake and assistance from local cooperators to make an appropriate determination.

Table 11. Lake eutrophication WQS for aquatic recreation use assessments.

Ecoregion	TP (μg/L)	chl-*a* (μg/L)	Secchi (m)
Northern Lakes and Forest – Lake trout (Class 2A)	< 12	< 3	> 4.8
Northern Lakes and Forest – Stream trout (Class 2A)	< 20	< 6	> 2.5
Northern Lakes and Forest – Aquatic Rec. Use (Class 2B)	< 30	< 9	> 2.0
North Central Hardwood Forest – Stream trout (Class 2A)	< 20	< 6	> 2.5
North Central Hardwood Forest – Aq. Rec. Use (Class 2B)	< 40	< 14	> 1.4
North Central Hardwood Forest – Aq. Rec. Use (Class 2B) Shallow lakes	< 60	< 20	> 1.0
Western Corn Belt Plains & Northern Glaciated Plains – Aq. Rec. Use (Class 2B)	< 65	< 22	> 0.9
Western Corn Belt Plains & Northern Glaciated Plains – Aq. Rec. Use (Class 2B) Shallow lakes	< 90	< 30	> 0.7

IX. Protection of waters used for the production of wild rice

Minn. R. 7050.0224 provides a sulfate standard of 10 mg/L to protect “water used for the production of wild rice during periods when rice may be susceptible to damage by high sulfate levels.” In March 2021, EPA disapproved Minnesota’s decision not to identify and include water that exceeds this standard on the impaired waters list. EPA subsequently proposed to include several waters as impaired. For this assessment and listing cycle, MPCA drew from EPA’s analysis and other components of the Guidance to develop an assessment methodology for the wild rice sulfate standard. The MPCA anticipates the further evolution of this methodology over future assessment and listing cycles.

Waters used for the production of wild rice are considered impaired if the average annual sulfate concentration exceeds, with statistical significance, the state water quality standard of 10 mg/L.

Average annual concentration is the measure that most closely follows the work that provided the basis for the current 10 mg/L sulfate standard and that most accurately captures the way that sulfate affects wild rice. The assessment methodology takes into account both data quantity and data variability by using a statistical test to provide a quantifiable high degree of confidence that the calculated average from the data adequately represents the actual average in the water.

- Waters used for the production of wild rice are, at minimum, those listed directly or indirectly in Attachment 2 of the Statement of Need and Reasonableness of the MPCA’s 2017 proposed wild rice sulfate standard rule (found at https://www.pca.state.mn.us/sites/default/files/wq-rule4-15j.pdf), and modifications made during the public comment period on that rulemaking. All such waters that have sulfate data meeting the criteria below are assessed.
- Assessments require data sets of at least five independent observations in the most recent 10 years that meet necessary QA/QC requirements and give an unbiased representation of overall conditions through the year.
    - Independent observations are samples taken at unique locations and days.
    - Unbiased representation preferably includes observations taken during different months of the year, over the time in which wild rice grows (e.g. eight samples taken over two years but all samples were taken in the month of May do not give an unbiased representation of overall conditions.
- Multiple measurements taken at the same site on the same day are treated as repeated measures and averaged. For measurements taken at multiple sites on the same day within a waterbody, the maximum value is used.
- Measurements taken in lakes at a depth greater than 3 meters are not used because wild rice tend to not grow in deep water.
- The average sulfate concentration is compared to 10 mg/L, using a specific level of statistical significance.
    - If the average sulfate concentration is greater than 10 mg/L (the lower confidence limit is greater than 10), the wild rice use is not supported and the water body is impaired.
    - If the average sulfate concentration is less than 10 mg/L (the upper confidence limit is less than 10), the wild rice use is fully supported.
    - If the average sulfate concentration is not significantly different than 10 mg/L, the assessment is considered inconclusive and the water is identified as needing additional monitoring and assessment.

- Determinations of statistical significance are made at an 80 percent confidence level, using the Kaplan-Meier estimator and a boot-strapped confidence interval. The choice of an 80% confidence interval conservatively balances the risk and cost of incorrectly listing a water as impaired, thus requiring a TMDL and corrective action, with the risk and cost of failing to list a water that is in fact impaired. If additional monitoring and assessment done subsequent to a listing shows the use is supported, this will lead to the correction of the incorrect listing.

As with all water-impairment assessments, best professional judgment that considers all relevant factors and evidence may be necessary and determinative in final listing decisions.

X. Protection of limited resource value waters (Class 7)

Limited resource value waters include surface waters of the state that have been subject to a use attainability analysis and have been found to have limited value as a water resource. These waters are specifically listed in rule (Minn. R. 7050.0470) and are protected so as to allow secondary body contact use, to preserve the groundwater for use as a potable water supply, and to protect aesthetic qualities of the water.

Standards (in Minn. R. 7050.0227) for limited resource value waters include the following:

- *Escherichia (E.) coli*: Not to exceed 630 organisms per 100 mL as a geometric mean of not less than five samples representative of conditions within any calendar month, nor shall more than 10% of all samples taken during any calendar month individually exceed 1260 organisms per 100 mL. The standard applies between May 1 and October 31. Assessment methodology is described in detail in Section VIII.A.
- Dissolved oxygen: At concentrations which will avoid odors or putrid conditions or at concentrations not less than 1 mg/L as a daily average, provided that measurable concentrations are present at all times.
- pH: minimum value of 6.0, maximum value of 9.0.
- Toxic pollutants not allowed in such quantities or concentrations that will impair the specified uses.

Application of toxic standards to Class 7 waters for assessment purposes includes applying the Maximum Standard for most pollutants or 100 times the Chronic Standard (CS), whichever is lower (Minn. R. 7050.0222, subp. 7, item E). However, for bioaccumulative pollutants the CS would apply. Because Class 7 waters may be used by game fish for spawning and/or maintaining minnow populations during brief periods in the spring, a special protection against bioaccumulative pollutants is needed.

XI. Removal of water bodies from the 303(d) Impaired Waters List

There are four ways in which water bodies are removed from the 303(d) List:

A. New and reliable data or information indicates that the water body is now meeting WQS.
B. A TMDL plan for reducing the sources of pollution is completed and approved by the EPA.
C. The sources of impairment are determined to be not caused by a pollutant or natural background conditions.
D. A correction to the list is required after it was determined that a water body was placed on the list in error, or reassessment with new standards or assessment methods does not indicate impairment.

It is important to note that in scenarios B and C above, the water body is still impaired and still appears on the Impaired Waters Inventory (until such time as the water body supports all its beneficial uses), but because a TMDL study is completed or not required. The following paragraphs provide more details on the four scenarios above.

A. Water body no longer impaired

In general, water body listing or delisting decisions will be made using the methods described in this Guidance. In practice, there will usually be more data available for the “delisting” assessment than was available for the “listing” assessment. New and old data will be considered together in the reassessments, unless tangible improvements of sufficient dimension to change impairment status have taken place in the reach, in which case only new data will be used in the delisting assessment. Improvements could include implementation of best management practices to reduce nonpoint sources, improvements in wastewater treatment, or some combination of nonpoint and point source reductions. If the new data show the water body to be un-impaired, the MPCA will recommend that the water body be delisted.

All delisting decisions are subject to review by the appropriate watershed assessment and professional judgment teams (see Section III) or the delisting committee for waters outside of the watersheds being assessed that year. Information about watershed improvements should be brought to the watershed assessment and professional judgment team or delisting committee for consideration. The MPCA will make a final determination on whether a water body can be considered no longer impaired, and should be submitted to the EPA for delisting.

It is essential that data used in the delisting assessment be collected under appropriate conditions. For DO and for pollutants with toxicity- and human health-based WQS, data should be from observations taken during critical conditions, i.e., those conditions most likely to result in exceedances of the standard. For example, if a water body was listed as impaired because of low DO, the measurements used to support delisting would likely need to be collected in the early morning (generally no later than 9:00 a.m., so as to reflect the daily minimum) during periods of very low flow. For other pollutants, data should be from observations that provide an accurate representation of the overall period of time under consideration and are not biased by, for example, being collected only during a certain season or under certain flow conditions.

The following is a summary of the specific data and assessment requirements needed to consider removing a water body from the 303(d) List, impaired because of exceedances of numeric standards:

Dissolved oxygen, pH, and total suspended solids must have:

- At least 20 observations in the most recent 10 years, of which at least 10 observations are in the most recent 5 years or at least 20 new observations in the most recent 5 years.
- Monitoring for new observations has occurred at times or under situations where exceedances of the WQS would be most likely to occur.
- Fewer than 10% of observations exceed the WQS.

River eutrophication must have:

- The causative variable (TP) and the response variable(s) that were used to list the AUID meet the standard.
- A minimum of 12 paired samples over a minimum of 2 years for total phosphorus, chlorophyll-*a*, and/or biochemical oxygen demand.
- A minimum of 20 pH samples over a minimum of 2 years.
- A minimum of 2 DO sonde deployments; each with a length of a minimum of 4 days and occurring in separate years during a similar index period to the listing deployment within the assessment window.

Pollutants toxic to aquatic life and drinking water nitrate must have:

- Sufficient ambient water quality monitoring to show, with reasonable certainty, that toxic pollutant concentrations no longer exceed the criteria for impairment and/or
- Evidence that the source of the toxic pollutant is no longer a source.
- The criterion for delisting toxic pollutants is essentially a determination that the impairment no longer exists. The monitoring required for this can vary significantly, depending on the pollutant and the situation. The criterion for impairment is strict and requires only two exceedances of the chronic standard within any three-year period or one exceedance of the maximum standard. A showing that exceedances are not occurring on even such an infrequent basis requires either a good deal of monitoring or monitoring at times and under situations where exceedances would be most likely to occur. As such, the delisting determination will inevitably require knowledge of the specific pollutant and the specific situation as well as significant professional judgment.

Fish contaminants must have:

- Five or more fish of the same species causing the impairment.
- A minimum of two years of data since the year the lake or river was added to the impaired waters list.
- Most recent data must show all fish species collected are not exceeding the threshold for impairment.
- The data show a downward trend in the annual 90th percentiles.
- For mercury, concentrations for a specific water body, species, and year has a 90th percentile less or equal to than 0.2 mg/kg (ppm).
- For PCBs and PFOS, MDH's fish consumption guidance has been removed or reduced to less restrictive than a meal per month and arithmetic mean concentration is less than 0.22 ppm for PCBs or 50 µg/kg (ppb) for PFOS. In addition, some PFOS-contaminated sites have a site-specific criterion of 0.37 µg/kg (ppb) for PFOS and they are identified in https://www.pca.state.mn.us/sites/default/files/wq-s6-61b.pdf.

***E. coli* bacteria must have:**

- At least 15 observations over a two-year period in the most recent 10 years.
- A minimum of 5 values per month for at least 3 months when the standard is applicable April – October, but preferably between June and September; data are combined for each month over most recent 10 years, unless there are a sufficient number of observations to aggregate data by month over consecutive 2-year time periods, or to calculate individual monthly or 30-day geometric means.
- A minimum of 5 values per month for at least 3 months when the standard is applicable April – October, but preferably between June and September; data are combined for each month over most recent years since corrective actions were taken in the watershed of sufficient dimension to change impairment status, unless there are a sufficient number of observations to aggregate data by month over consecutive 2-year time periods, or to calculate individual monthly or 30-day geometric means.
- In either case, no exceedance of the monthly mean standard (126 organisms per liter) by the geometric mean in any of those months for 10-year aggregated data or less than 10% of months exceed the standard for 2-year aggregated or individual monthly or 30-day geometric means.
- In either case, fewer than 10% of sample observations exceed "maximum" standard (126 organisms per liter).

Lake eutrophication must have:

- At least 8 paired total phosphorus (TP), corrected chl-*a*, and Secchi measurements (June to September) over a minimum of 2 years for the most recent 10 years.
- If TP meets the standard, and either chl-*a* or Secchi meet the standard, the lake will be delisted.
- If TP exceeds the standard and corrected chl-*a* **and** Secchi meet the standard, and an improving trend in TP is observed or management activities are in place to maintain improved chl-*a* or Secchi observations, the lake may be delisted. This will require the local entity to provide information that details how the response conditions will be met over time.

Biological indicators should have:

- New data from the original listing station(s) indicating conditions are now supporting of aquatic life.
- An evaluation of any new biological data and other lines of evidence considered comprehensively, including upstream/downstream conditions, do not contradict a finding of full support.
- An evaluation that any stressors to the biology that may have been previously identified as part of the TMDL process indicate measured improvement.

Water bodies with impaired aquatic communities can be delisted utilizing the same criterion as listing (Section V. B) if additional bio-monitoring indicates that the community is no longer impaired when compared to the IBI threshold (±confidence interval). Overall assessment of whether an AUID adequately supports aquatic life involves the review of the parameter-level evaluations and data quality in conjunction with all available supporting information (flow, habitat, precipitation, etc.) to make an overall use-support determination. For a given AUID, there may be chemistry indicator data, biological indicator data, or both types of data available for assessment. The final assessment takes into consideration the strength of the various indicators and the quality of the data sets and, in addition, looks at upstream and downstream conditions to gain a better understanding of the interactions between the individual AUID and the larger water body and watershed.

B. EPA-approved TMDL plan

The most common way waters are removed from the 303(d) List is through the completion of the TMDL study. Under the current federal TMDL regulation, the TMDL process must progress through the step where an EPA-approved plan is in place that indicates in general how the river reach or lake is to be brought back into compliance with WQS. That is, under current EPA regulations, the water body does not need to be brought back to an un-impaired condition to be delisted. Irrespective of this EPA regulation, the MPCA is committed, with the help of local entities, to improving the water quality in all impaired waters so beneficial uses are restored, where restoration is possible. To that end, an AUID that has an approved TMDL plan for a pollutant no longer appears on the 303(d) List, but it remains on the Inventory of Impaired Waters with a 4A category until it is found to be no longer impaired.

C. Water body impaired because of a non-pollutant or natural background conditions

A water body may be removed from the 303(d) list after it was determined that there are only non-pollutant sources contributing to the impairment. These sources might include changes to the water body such as dams, impoundments or other anthropogenic factors affecting stream connectivity or flow. These impairments remain on the Impaired Water List with a 4C Category.

If it is determined that an impairment is due to natural background conditions, that impairment can be removed from the 303(d) list but remain on the Impaired Waters List as with a Category of 4D. Examples of 4D impairments include shallow northern Minnesota lakes naturally higher in nutrients than current deep-lake WQSs, and rivers influenced by wetlands which contribute to naturally low dissolved oxygen.

D. List correction

If a water body was placed on the list in error either by incorrect data, or would not have been placed on the list under current standards or methodology, the reach will be removed from the list as a correction.

XII. Sources of information and MPCA contacts

The readers of this document are encouraged to access the sources of information listed in this section. Included are email addresses and phone numbers of MPCA staff that work in areas relevant to the protocols and procedures in this Guidance. They are listed alphabetically by subject area. Also provided are some pertinent websites, listed by agency.

1. 303(d) List, Inventory of Impaired Waters, general questions and comments: Miranda Nichols at miranda.nichols@state.mn.us or 651-757-2614.
2. 305(b) integrated report: Miranda Nichols at miranda.nichols@state.mn.us or 651-757-2614.
3. Basin or watershed planning questions: Glenn Skuta at glenn.skuta@state.mn.us or 651-757-2730.
4. Biological impairment: Scott Niemela at scott.niemela@state.mn.us or 218-828-6076.
5. Citizen monitoring programs: clmp@state.mn.us (lakes), csmp@state.mn.us (streams) or 800-296-6300.
6. Effluent limits for toxic pollutants:
   Dann White dann.white@state.mn.us or 651-757-2820.
7. Fish consumption guidance: Minnesota Department of Health at 800-657-3908.
8. Lake and river eutrophication methodology: Lee Engel at lee.engel@state.mn.us or 651-757-2339.
9. Limited Resource Value Waters (Class 7): Carol Sinden at carol.sinden@state.mn.us or 651-757-2727.
10. TMDL process, general questions and comments: Celine Lyman at celine.lyman@state.mn.us or 651-757-2541.
11. Data management and water quality data for specific water bodies: May Knight at mary.knight@state.mn.us or 651-757-2424.
12. Water quality standards: Angela Preimesberger at angela.preimesberger@state.mn.us or 651-757-2656.

All MPCA staff can be reached toll free at 800-657-3864 or 651-296-6300 in the Twin Cities Metropolitan Area.

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XIV. Appendices

Appendix A. State overall and beneficial use reporting categories

Category	Description
2	Waterbody's assessed designated uses are fully supported, the designated use is fully supported, or parameter meets standards.
3	Data insufficient or inconclusive to assess.
4A	Impaired and a TMDL study has been approved by EPA.
4B	Impaired but a TMDL study is not required because water quality standards are expected to be met in the near future.
4C	Impaired but a TMDL study is not required because the impairment is not caused by a pollutant.
4D	Impaired but a TMDL study is not required because the impairment is due to natural conditions with insignificant anthropogenic influence.
4E	Impaired but existing data strongly suggests a TMDL study is not required because impairment is not caused by a pollutant or is due to natural conditions; a final category determination will be made pending confirmation from additional data collection.
5	Impaired and a TMDL study has not been approved by EPA.

Appendix B. Minnesota's TMDL priorities

The MPCA has prioritized Total Maximum Daily Loads (TMDLs) for the years 2016-2022 as part of EPA's Long-Term Vision for Assessment, Restoration, and Protection under the Clean Water Act Section 303(d) Program. These TMDL priorities are a subset of identified by the TMDL target completion dates on the 303(d) list. Minnesota's TMDL priorities identified for the prioritization goal of EPA's Long-Term Vision are those water bodies listed for conventional pollutants with an estimated TMDL target completion date of 2021 or earlier. Water bodies listed for nonconventional pollutants (chloride and mercury for example) will continue to be done through a separate process rather than through the watershed approach. A small number of water bodies listed for conventional pollutants have been deferred to later dates when Cycle 2 of the watershed approach is in progress. For the entire TMDL Priority Framework Report, go to the MPCA's TMDL policy and guidance webpage at https://www.pca.state.mn.us/water/tmdl-policy-and-guidance. The MPCA is currently working on priorities through 2032.

Watershed Restoration and Protection Strategy (WRAPS)

WRAPS reports will be done on a 10-year watershed cycle and the TMDLs for conventional pollutants in those watersheds will be done as part of the WRAPS process, with some exceptions (see deferred TMDLs below). The conventional pollutants are DO, pH, temperature, turbidity, TSS, bacteria, ammonia, nitrates, nutrients, and biological impairments.

The State of Minnesota has adopted a watershed approach to address the state's 80 major watersheds (denoted by 8-digit hydrologic unit code or HUC). This watershed approach incorporates water quality assessment, watershed analysis, civic engagement, planning, implementation, and measurement of results into a 10-year cycle that addresses both restoration and protection. The Watershed Restoration and Protection Strategy (WRAPS) report is done as a result of that work. In addition to the WRAPS report, a watershed TMDL study is done.

As part of the watershed approach, waters not meeting state standards are still listed as impaired and studies are performed, as they have been in the past, but in addition the watershed approach process facilitates a more cost-effective and comprehensive characterization of multiple water bodies and overall watershed health. A key aspect of this effort is to develop and utilize watershed-scale models and other tools to identify strategies and actions for point and nonpoint source pollution that will cumulatively achieve water quality targets. For nonpoint source pollution, this report informs local planning efforts, but ultimately the local partners decide what work will be included in their local plans. The WRAPS report provides a HUC8-level strategy to address the impairments and protection needs in the watershed and allows for local partners to develop more specific plans at the local level. The Section 319 Small Watersheds program is working with local governments to develop very detailed nine element plans on a smaller scale to qualify for Section 319 grant funding.

Appendix C. Sources of data used for assessment

Involvement of local units of government and other governmental agencies in the monitoring of water quality is always encouraged, and the MPCA actively seeks data from all sources utilizing appropriate QA/QC with annual calls for data.

Analytical labs providing data must be certified under the lab certification program operated by MDH, and the data to be used in assessments should be entered into the MPCA's ambient water quality database, EQuIS (Environmental Quality Information System). A major aspect of monitoring that the MPCA must consider when reviewing data for use in assessments is the purpose for which the data were collected. For example, samples collected to characterize "events" such as the effects of storm runoff on a river may not be suitable, if used alone, to characterize the overall water quality of the river.

Data from any source that has been entered in EQuIS or another MPCA database, reviewed, and found to satisfy QA/QC requirements will be considered for use in assessments. Major examples include:

- Data from any source submitted to MPCA for entry into the agency's ambient water quality monitoring database, EQuIS (https://www.pca.state.mn.us/data/environmental-quality-information-system-equis). This includes, but is not limited to, data collected by:
  - MPCA monitoring programs,
  - Projects funded by state or federal money (e.g., Clean Water Partnership or National Lake Assessment Program data),
  - Any organization sending data to a lab contracted with MPCA to submit data to EQuIS,
  - MPCA's citizen monitoring programs, and
  - Minnesota Department of Agriculture's water quality data.
- Continuous water quality data (e.g., flow, DO, temperature collected internally or by parties outside the MPCA) accessible through the MPCA/DNR's shared database for continuous data.
- Biological data, specifically IBI scores from MPCA's internal biological database.

Data obtained through projects the MPCA funds must be the result of a clearly defined and documented purpose and it must satisfy specific data needs. This documentation is called an "information protocol," and it has proven to be very useful to MPCA staff considering the broad range of types and purposes of monitoring programs carried out by agencies and other organizations.

The MPCA may also search out data from sources not amenable to EQuIS entry. Sources of water quality data outside the MPCA that are considered each year for use in water quality assessments include:

- Neighboring states and tribes in Minnesota, only if found at https://www.waterqualitydata.us/.
- Metropolitan Council Environmental Services, found at www.eims.metc.state.mn.us.
- United States Geological Survey, found at https://www.waterqualitydata.us/.
- If applicable, the Upper Mississippi River Restoration Program Long Term Resource Monitoring, found at https://umesc.usgs.gov/ltrm-home.html.

Appendix D. Lake, shallow lake, and wetland differentiation

Some of the factors used to separate lakes, shallow lakes, and wetlands are as follows:

Factor	Lakes	Shallow lakes	Wetlands
Protected Waters Inventory (PWI) Code	Typically coded as "L or LP" in PWI	May be coded as either "L, LP or LW" in PWI	Typically coded as a "LW" in PWI
Depth, maximum	Typically >15 feet	Typically < 15 feet	Typically < 7 feet
Littoral area	Typically <80%	Typically >80%	Typically 100%
Area (minimum)	Typically > 10 acres (NDH)	Typically > 10 acres (NDH)	No minimum
Thermal stratification (summer)	Stratification common but dependent upon depth, size and fetch	Typically do not thermally stratify	Typically do not stratify.
Fetch*	Significant fetch depending on size & shape	Fetch is variable depending on size & shape	Rarely has a significant fetch
Substrate	Consolidated sand/silt/gravel	Consolidated to mucky	Mucky to unconsolidated
Shoreline features	Generally wave formed, often sand, gravel or rock	Generally wave formed, often sand, gravel or rock	Generally dominated by emergents
Emergent vegetation & relative amount of open water*	Shoreline may have ring of emergents; vast majority of basin open water.	Emergents common, may cover much of fringe of lake; basin often has high percentage of open water.	Emergents often dominate much of basin; often minimal open water.
Submergent vegetation	Common in littoral fringe, extent dependent on transparency	Abundant in clear lakes; however may be lacking in algal-dominated turbid lakes.	Common unless dominated by an emergent like cattail.
Dissolved Oxygen	Aerobic epilimnion; hypolimnion often anoxic by midsummer	Aerobic epilimnion but wide diurnal flux possible	Diurnal flux & anaerobic conditions common
Fishery	Typically managed for a sport/game fishery. May be stocked. DNR fishery assessments typically available.	May or may not be managed for a sport fishery. If so, fishery assessment should be available. Winter aeration often used to minimize winterkill potential.	Typically not managed for a sport fishery. Little or no DNR fishery information. Seldom aerated. May be managed to remove fish & promote waterfowl.
Uses	Wide range of uses including boating, swimming, skiing, fishing; boat ramps & beaches common	Boating, fishing, waterfowl production, hunting, aesthetics; limited swimming; may have boat ramp, beaches uncommon	Waterfowl & wildlife production, hunting, aesthetics. Unimproved boat ramp if any. No beaches.

* Fetch and open water play a large role in these determinations.

Appendix E. Assessing and communicating the quality of waters that occur wholly or partially within federally recognized Indian reservations

Goal: Work with tribes to monitor, assess, and communicate the quality of waters that are within, or partially, within the boundaries of Indian reservations.

Background: Measuring and communicating water quality is core to the mission of the MPCA. The Clean Water Act requires delegated programs to determine if waters are meeting standards designed to protect uses like fishing and swimming.

Waters that do not meet water quality standards (WQS) are designated as "impaired." Delegated programs are required to submit a draft list of impaired waters to the U.S. Environmental Protection Agency (EPA) for approval every two years and establish a TMDL of pollutants that, if met, will result in attaining the standards. In addition to federal law requirements, Minnesota state law[1] requires the MPCA to determine if any waters of the state are impaired.[2] Because of the broad definition of waters of the state, the state's impaired waters requirement applies to more waterbodies than the federal requirements. Therefore, the MPCA includes a state-only impaired waters list with the required federal list in order to have a comprehensive listing of impaired waters within Minnesota. Given that people across the state use waters, this establishes a common approach for communicating about the condition of waters wholly or partially within reservation boundaries.

The Grand Portage Band and Fond du Lac Band have received EPA delegation, known as "treatment as a state" (TAS), to establish WQS and have adopted water quality standards that have been approved by EPA. Several other Bands are in the process of applying for TAS for water quality standards. The Leech Lake Band has submitted their TAS application to EPA. EPA public noticed the application on May 31, 2019 and took comments through July 15, 2019. A final decision is pending, but MPCA anticipates that Leech Lake will also receive TAS status. On October 26, 2016, EPA adopted regulations to establish a process for eligible tribes to obtain TAS to list waters within their reservations as impaired under section 303(d) and establish TMDLs.

The MPCA recognizes that both states and tribes (whether or not they have obtained TAS for the WQS program or the 303(d) program) are invested in protecting and restoring all waters. The MPCA also recognizes that EPA has stated that its approval of the State's 303(d) impaired waters list does not extend to waters within Indian reservations,[3] including fee and parcels held in trust (tribal trust lands), and that EPA will take no action to approve or disapprove the list with respect to waters within Indian reservations for purposes of section 303(d). The MPCA also recognizes recent (June 30, 2021) comments from the Minnesota Indian Affairs Council that object to MPCA's general practice of including tribal waters on the list, and identify inconsistent efforts for tribal engagement.

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[1] *Minn. Stat. §§ 115.03, subd . 1 and 115.44; Minn. Laws 2005, 1st Sp.1, ch. 1, art. 2, § 151; and Minn. R. 7050.0150 (impaired waters authority). See also Minn. Stat. ch. 1140 and Minn. R. 7052.0200 (TMDL authority).*

[2] *Minn. Stat. § 115.01, subd. 22. "Waters of the state" means all streams, lakes, ponds, marshes, watercourses, waterways, wells, springs, reservoirs, aquifers, irrigation systems, drainage systems and all other bodies or accumulations of water, surface or underground, natural or artificial, public or private, which are contained within, flow through, or border upon the state or any portion thereof.*

[3] *Language from EPA's Approval Letter of MPCA's 2018 Impaired Waters List "EPA's approval of Minnesota's Section 303(d) list extends to all water bodies on the list with the exception of those waters that are within Indian Country, as defined in 18 U.S.C. § 1151. EPA is taking no action to approve or disapprove the State's list with respect to those waters at this time. EPA, or eligible Indian Tribes, as appropriate, will retain responsibilities under CWA Section 303(d) for those waters".*

The MPCA's watershed approach to monitoring includes notification to tribes and offers to collaborate to develop a mutually agreeable monitoring plan for waters that occur wholly or partially within their reservation boundaries. Following two years of watershed monitoring, MPCA scientists strive to work with local resource managers, including tribal staff familiar with the monitoring efforts, to evaluate the data and determine if waters are meeting state WQS. The MPCA is working to improve inclusion of tribal water resources staff in the watershed assessment team portion of the process before the MPCA makes a draft impaired waters list available for public comment.

Approach for assessing and communicating water quality: The following represents the approach MPCA has taken in developing the 2022 impaired waters list, and generally plan to continue to take moving forward. However, MPCA will also consult with any tribe that wishes to discuss whether their waters should be included on the impaired waters list, and may make changes to the approach based on the outcome of that consultation.

1. MPCA will continue to work with tribes in advance of monitoring to agree on plans that include locations, parameters, roles, responsibilities and processes.
2. MPCA will share data with tribes and will engage tribes in the evaluation of monitoring data in light of applicable state WQS. The MPCA likewise appreciates tribes sharing their monitoring data.
3. MPCA will invite tribal water resources staff to discuss the assessment results with the MPCA's watershed assessment team – a discussion that occurs much prior to any public notice.
4. For waters deemed to be impaired that:
    a. Are partially within the boundaries of a federally recognized Indian reservation (but are not located wholly within a federally recognized Indian reservation, or serve as a border between a federally recognized Indian reservation and Minnesota land), the MPCA will include such waters on Minnesota's Impaired Waters List and include a footnote with each that states: "*This body of water is partially within a federally recognized Indian reservation and does not serve as a border between a federally recognized Indian reservation and Minnesota land. The state and tribe have worked cooperatively on this water quality assessment and agree that the water should be included on the State's impaired waters list. For the purposes of the 303(d) list, the assessment of the portion of the waterbody within the reservation is provided as information only to EPA because EPA does not approve the State's impaired waters listings for purposes of section 303(d) for waters within the boundaries of an Indian reservation. Note that the MPCA includes fee lands and parcels held in trust (tribal trust lands) in the definition of Indian reservation.*"
    b. Are located wholly within a federally recognized Indian reservation, the MPCA will send a list of these waters to EPA, separate from the impaired water list but accompanying the list and include the following statement in the title of this list: *"This list was prepared under authority in state law to determine whether waters within the state are impaired. The MPCA includes this state-only list in order to have a comprehensive list of impaired waters. For purposes of the 303(d) list, these assessments are provided as information only to EPA because these water bodies are located wholly within a federally recognized Indian reservation and EPA does not approve the State's impaired waters listings for purposes of section 303(d) for waters that are within the boundaries of an Indian reservation. Note that the MPCA includes fee lands and parcels held in trust (tribal trust lands) in the definition of Indian reservation."*
5. Prior to putting the draft Impaired Waters List on public notice, the MPCA will communicate with tribes waters that are partially or wholly within reservation boundaries and determined to be impaired using state WQS. Such waters will be indicated with a footnote or included on the separate 'wholly' list as specified in #3 above.
6. MPCA and tribal representatives will discuss and determine whether there is a mutual desire to cooperatively develop restoration and protection strategies, including TMDLs, for impaired waters that are partially or wholly within reservation boundaries.

Appendix F. Supplemental information on biological assessment in Minnesota

Basis for assessment of biological community

Assessment of the biological community for impairment is based on the narrative water quality standards (WQS) and assessment factors in Minn. R. 7050.0150. The most relevant part, Minn. R. 7050.0150, subp. 6 is quoted below:

> *Subp. 6. Impairment of biological community and aquatic habitat. In evaluating whether the narrative standards in subpart 3, which prohibit serious impairment of the normal aquatic biota and the use thereof, material alteration of the species composition, material degradation of stream beds, and the prevention or hindrance of the propagation and migration of aquatic biota normally present, are being met, the commissioner will consider all readily available and reliable data and information for the following factors of use impairment:*
>
> *A. An index of biological integrity calculated from measurements of attributes of the resident fish community, including measurements of:*
> *1) species diversity and composition;*
> *2) feeding and reproduction characteristics; and*
> *3) fish abundance and condition.*
>
> *B. An index of biological integrity calculated from measurements of attributes of the resident aquatic invertebrate community, including measurements of:*
> *1) species diversity and composition;*
> *2) feeding characteristics; and*
> *3) species abundance and condition.*
>
> *C. An index of biological integrity calculated from measurements of attributes of the resident aquatic plant community, including measurements of:*
> *1) species diversity and composition, including algae; and*
> *2) species abundance and condition.*
>
> *D. A quantitative or qualitative assessment of habitat quality, determined by an assessment of:*
> *1) stream morphological features that provide spawning, nursery, and refuge areas for fish and invertebrates;*
> *2) bottom substrate size and variety;*
> *3) variations in water depth;*
> *4) sinuosity of the stream course;*
> *5) physical or hydrological alterations of the stream bed including excessive sedimentation;*
> *6) types of land use in the watershed; and*
> *7) other scientifically accepted and valid factors of habitat quality.*
>
> *E. Any other scientifically objective, credible, and supportable factors.*

A finding of an impaired condition must be supported by data for the factors listed in at least one of items A to C. The biological quality of any given surface water body will be assessed by comparison to the biological conditions determined by the commissioner using a biological condition gradient model or a set of reference water bodies which best represents the most natural condition for that surface water body type within a geographic region.

Additional language supporting the use of narrative WQS in wetlands is found in Minn. R. 7050.0222, subp. 6, which defines the protection of Class 2D waters (wetlands) as follow:

> *"The quality of Class 2D wetlands such as to permit the propagation and maintenance of a healthy community of aquatic and terrestrial species indigenous to wetlands, and their habitats. Wetlands also add to the biological diversity of the landscape. These waters shall be suitable for boating and other forms of aquatic recreation for which the wetland may be usable. This class of surface water is not protected as a source of drinking water. …"*

In addition to the narrative language in rule, which supports assessment of biological communities and habitat, Minnesota rules also include numeric biological criteria for assessment of fish and macroinvertebrates in streams and rivers. These biocriteria are found in Minn. R. 7050.0222, subps. 2d, 3d, and 4d (Table 12). This rule language includes biocriteria values for both fish and macroinvertebrates, for different stream types and TALUs. Supporting documentation incorporated by reference into rule for these biocriteria are found in Minn. R. 7050.0222, subps. 2c, 3c, and 4c. These documents include fish and macroinvertebrate data collection protocols, IBI calculation, BCG model development, and biocriteria development for streams.

Table 12. Tiered aquatic life use (TALU) numeric biological criteria for the assessment of fish and macroinvertebrate communities in rivers and streams using the index of biological integrity or IBI.

	Class	Class Name	Use Class	General (g) Use IBI Threshold	Exceptional (e) Use IBI Threshold	Modified (m) Use IBI Threshold	90% Confidence Limit (±)
Fish IBI Classes	1	Southern Rivers	2B	49	71		11
	2	Southern Streams	2B	50	66	35	9
	3	Southern Headwaters	2B	55	74	33	7
	4	Northern Rivers	2B	38	67		9
	5	Northern Streams	2B	47	61	35	9
	6	Northern Headwaters	2B	42	68	23	16
	7	Low Gradient	2B	42	70	15	10
	10	Southern Coldwater	2A	50	82		13
	11	Northern Coldwater	2A	35	60		10
Macroinvertebrate IBI Classes	1	Northern Forest Rivers	2B	49	77		10.8
	2	Prairie Forest Rivers	2B	31	63		10.8
	3	Northern Forest Streams RR	2B	53	82		12.6
	4	Northern Forest Streams GP	2B	51	76	37	13.6
	5	Southern Streams RR	2B	37	62	24	12.6
	6	Southern Forest Streams GP	2B	43	66	30	13.6
	7	Prairie Streams GP	2B	41	69	22	13.6
	8	Northern Coldwater	2A	32	52		12.4
	9	Southern Coldwater	2A	43	72		13.8

The aquatic life use-support assessment methodology described in this Guidance fully supports the narrative and numeric standards in Minnesota rule and protects the biological integrity of rivers, streams, and wetlands by:

- Measuring attainment directly through sampling of the aquatic biota.
- Controlling biological and sampling variability through regionalization, classification and strict adherence to sampling protocol.
- Establishing impairment thresholds based on data collected from reference (least-disturbed) waters of the same class.

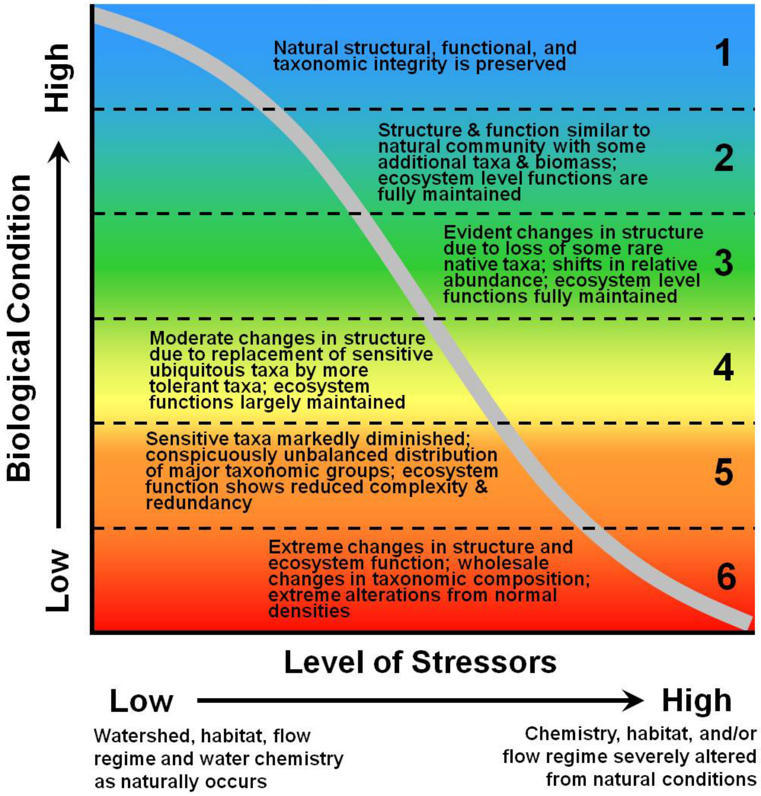
- Incorporating a confidence limit (based on the repeatability of the IBI) to account for variability within the aquatic community because of natural spatial and temporal differences and sampling or method errors.

Biological condition gradient

The Biological Condition Gradient (BCG) is a conceptual model of aggregated biological knowledge used to describe changes in biological communities along a gradient of increasing stress. This model is based on a combination of ecological theory and empirical knowledge. A number of indices have been developed to measure the biological condition in aquatic systems (e.g., IBI, RIVPACS; Karr et al. 1986, Hawkins et al. 2000, Whittier et al. 2007), but these measures are based on the available conditions that are used to develop the models. The BCG differs from these in that it provides a common "yardstick" of biological condition that is rooted in the natural condition. As a result, the BCG can be used to develop biocriteria that are consistent across regions and stream types in Minnesota. This is particularly important for a state such as Minnesota where the range of conditions are regionally distinct and extreme (i.e., relatively pristine to degraded). The BCG divides biological condition into six levels that are intended to be manageable and useful for water quality managers (see BCG model below). More detailed descriptions of the BCG can be found in EPA (2005) and Davies and Jackson (2006).

The development of the BCG models for warm water rivers and streams and lakes involved input from biological experts from the MPCA and DNR familiar with aquatic communities in Minnesota. BCG models were developed for fish and macroinvertebrates for each of the seven warm water stream classes and for four groups of lakes. A cold water BCG for streams was also developed and involved experts from Minnesota, Wisconsin, Michigan, and several tribes. In Minnesota, this included two classes each for fish and macroinvertebrates. Model development for each stream class involved reviewing biological community data from monitoring sites and then assigning that community to a BCG level (1-6). Similar model development was completed for lakes, utilizing the four lake groups. A sufficient number of samples were assessed to develop a model which can duplicate the panel's BCG level assignments. This model (Figure 3) was then used to assign BCG levels to all monitoring sites in MPCA's biological monitoring database for streams and MDNR's Lake Database for lakes.

Figure 3. Model used to assign BCG levels to Minnesota's biological monitoring sites.



Selection of reference sites for rivers and streams

Minnesota has developed an index to measure *a priori* the degree of human disturbance at a stream class called the Human Disturbance Score (HDS) (Table 13). The HDS includes both watershed and reach level measures of human disturbance which when combined have a maximum score of 81 (see Table 4. Metrics and scoring for Minnesota's Human Disturbance Score see Table 13 below). Reference sites were identified as those with an HDS score of 61 or greater (i.e., a 25% decline from the maximum score). Once sites were selected based on their HDS score, an additional filter was applied to remove sites disparately influenced by nearby stressors. All sites in close proximity to urban areas (site within or adjacent to urban area), feedlots (feedlot at or immediately upstream of site [only streams >50 mi^2]), or point sources (continuous point source <5 mi upstream of site) were removed. The remaining sites (i.e., those meeting the HDS threshold and meeting the proximity criteria) were considered to be minimally or least disturbed and therefore representative of attainment of Minnesota's aquatic life use goals. Reference sites were selected from each of the fish and macroinvertebrate classes and the 25th percentile of IBI scores was determined.

Table 13. Metrics and scoring for Minnesota's Human Disturbance Score.

Human Disturbance Score Metric	Scale	Primary Metric or Adjustment	Maximum Score
Number of animal units per sq km	watershed	primary	10
Percent agricultural land use	watershed	primary	10
Number of point sources per square km	watershed	primary	10
Percent impervious surface	watershed	primary	10
Percent channelized stream per stream km	watershed	primary	10
Degree channelized at site	reach	primary	10
Percent disturbed riparian habitat	watershed	primary	10
Condition of riparian zone	reach	primary	10
Number of feedlots per sq km	watershed	adjustment	-1
Percent agricultural land use on >3% slope	watershed	adjustment	-1
Number of road crossings per sq km	watershed	adjustment	-1 or +1
Percent agricultural land use in 100m buffer	watershed	adjustment	-1
Feedlot adjacent to site	reach (proximity)	adjustment	-1
Point source adjacent to site	reach (proximity)	adjustment	-1
Urban land use adjacent to site	reach (proximity)	adjustment	-1
		Maximum	81

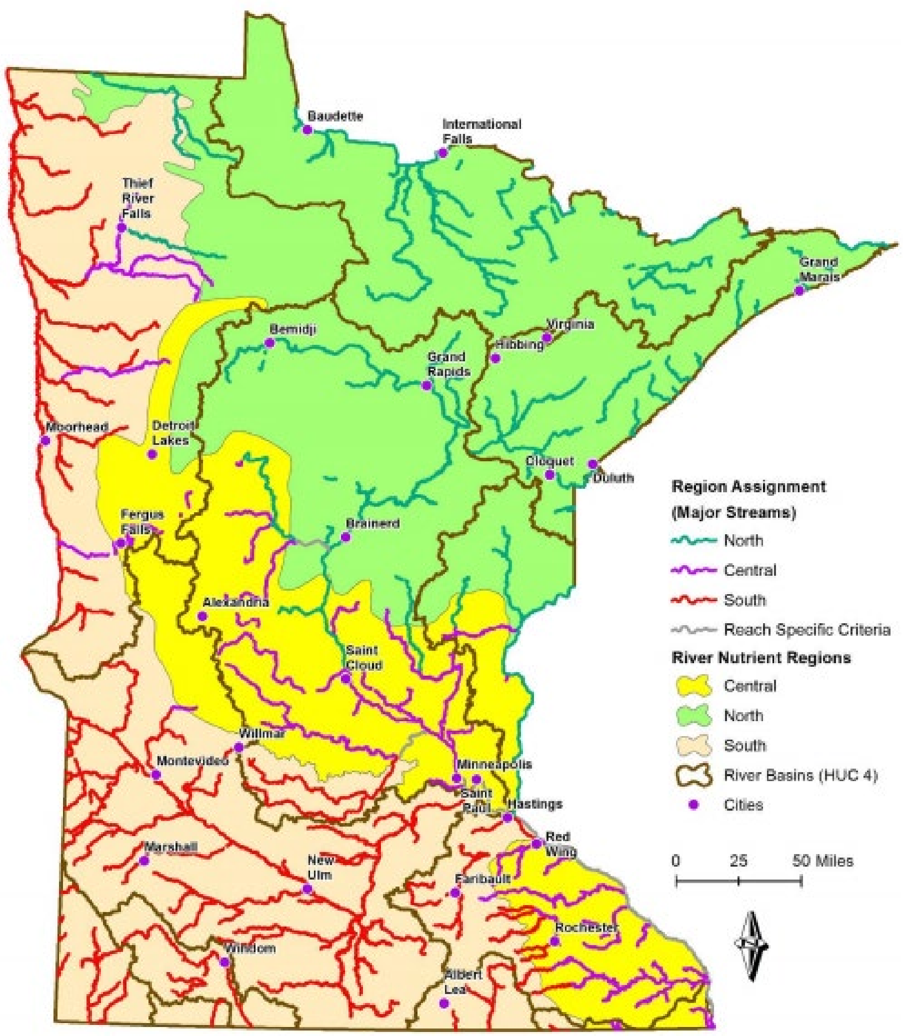
Appendix G. Supplemental information on river eutrophication assessment in Minnesota

The following information is intended to guide the completion of the RES assessments. This includes determination of the correct regional standard to apply, data requirements and summarization, and guidance for specific situations encountered during the assessments.

Assignment of regional standards

When an HUC-8 watershed is located wholly within a RNR (Figure 4), or where a vast majority of the watershed is within a single RNR, the RNR assignment is made to the dominant RNR. When a HUC-8 is characterized by multiple RNRs, a closer inspection was required and 11-digit HUCs were incorporated into the mapping coverage to allow for refinement of boundaries to determine the appropriate RNR assignment. In a few instances, where two 8-digit HUCs meet prior to entering the major mainstem river (e.g. North Fork and South Fork Crow Rivers) a site-specific standard was required and these reaches are noted on the RNR map. The MPCA will update the RNR map as needed; Heiskary and Parson (2013) provide further details on the mapping approach.

Figure 4. Statewide River Nutrient Region map.



During the assessment, the assigned RNR should be reviewed if there are questions regarding the AUID classifications when a river flows from one RNR to the next or where adjacent or upstream/downstream AUIDs have different RNR designations.

Minimum data requirements for total phosphorus (TP), chlorophyll-*a* (sestonic, corrected for pheophytin) or BOD_5 pH and periphyton chl-*a* (benthic, corrected for pheophytin)

The rule and the legal documents supporting and explaining the rule (SONAR Book 2, Minn. R. 7050, and Heiskary et. al. 2013) describe the following minimum data parameters:

- **Number of years**. Samples must be collected over a minimum of two years within the most recent 10-year time period (SONAR Book 2).
- **Time of year**. Data used for assessments must be collected from June to September (Minn. R. 7050).
- **Number of TP data points**. Based on a minimum of two years of monitoring, a minimum of six individual data points per summer for the causative variable TP must be collected (as noted in SONAR Book 2, pp. 81).
- **Response variables chlorophyll-a (chl-*a*), BOD5, pH**. In addition, the response variables chl-*a* or BOD5 or pH are collected concurrent with TP. A minimum of 12 measurements considering the above minimum data requirements for the 10-year assessment period are required for an assessment to be conducted (SONAR Book 2). While this minimum will typically be achieved over two years of sampling, it may also be achieved by multiple years (e.g. three years with four samples per year).

The term "representative" is used repeatedly in these definitions and implies that samples are to be collected across the summer season so they "represent" the entire season. Since river flow varies during individual summers and among summers, it is assumed samples will be collected over a range of flows; hence, the need to collect multiple samples over each summer and the need for two or more years of sample collection. While no specific flows are established for (or prohibited from) sample collection, the river must exhibit some amount of unidirectional flow for samples to be collected. If flows are so low that water is pooled or stagnant at the sample site and there is no evident downstream flow, these conditions must be documented and samples should not be utilized for river eutrophication assessment.

Data requirements specific to diel dissolved oxygen flux assessment

Diel DO flux is measured by means of probes (also referred to as a sonde) that are deployed for a minimum of four consecutive days in the river reach (AUID) being assessed. While these measures could be conducted at any time within the June through September timeframe, it is preferred that the measures be taken late summer from mid-July through August. Ideally, flows are relatively stable during the time the sonde is deployed. Due to interannual variability and the varied duration of single-year diel DO deployments, sonde deployments must meet the minimum deployment length and deployments must occur in a minimum of two summers in the assessment period to be considered representative of river conditions. Details on methods for collecting instrumented DO data for the calculation of diel DO flux are provided in technical support documents (Heiskary et al. 2013 and Heiskary and Markus 2003).

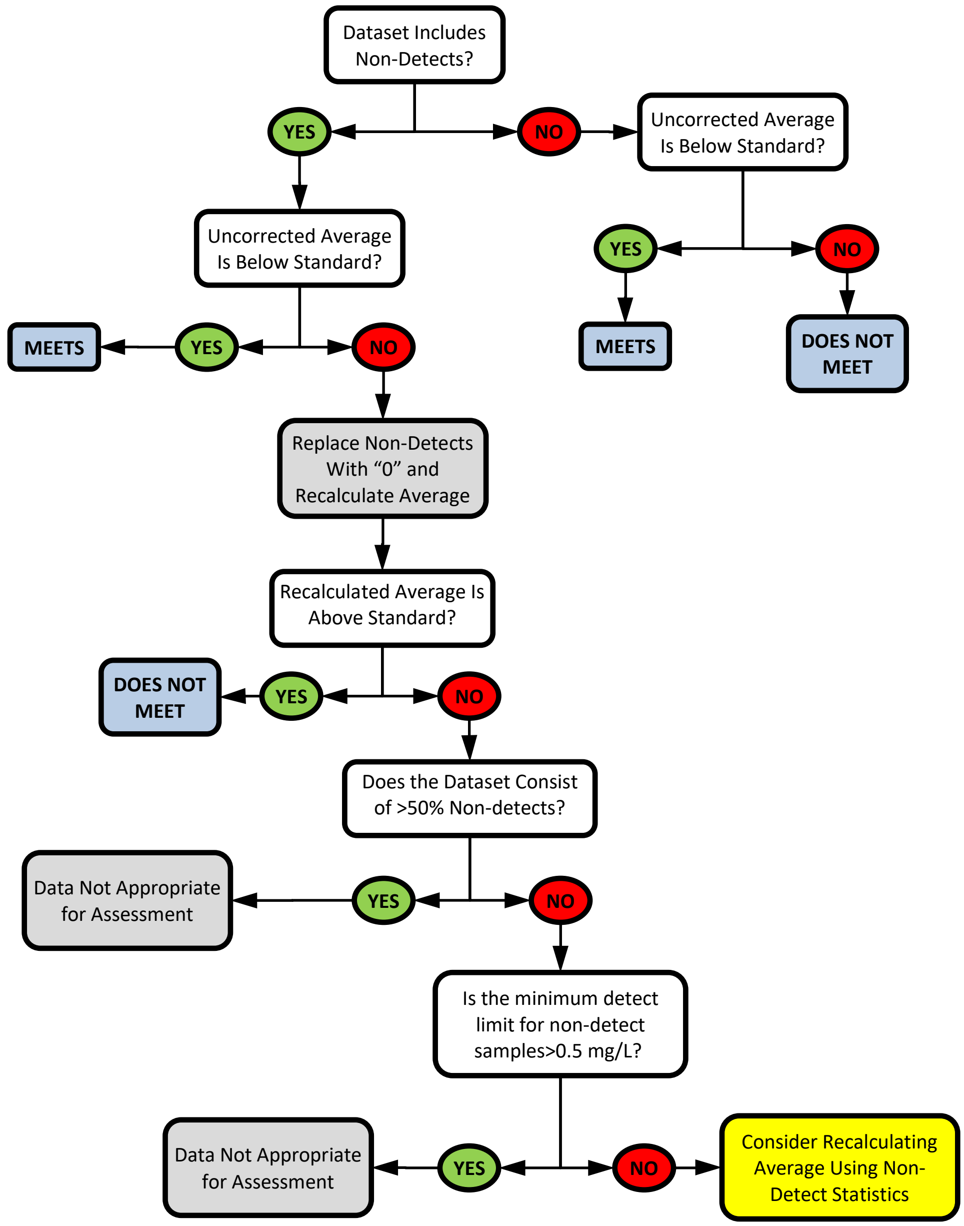
Determination of use assessment

The final step in assessment is determining if the RES has been met or exceeded for the water body based on the data collected. Minnesota's RES is a two-part standard involving a causative variable (TP) and response variables that indicate the presence of eutrophication (i.e., undesirable levels of sestonic or suspended algae, benthic or attached algae, or excessive rooted vegetation). For assessment purposes this means the cause indicator (TP) and response indicators (chl-*a*, BOD_5, diel DO flux, pH, or periphyton) are used in combination and not independently. The eutrophication rule clearly states the requirement that cause and at least one response indicators must both be exceeded to indicate a polluted condition.

Assessment staff should use the following information when assessing water bodies for the river eutrophication standard:

- **Primary and supplementary assessment statistics**. For chl-*a* and BOD5 data, as with TP data, summer-means for the entire 10-year assessment period are calculated from the available data and considered in the assessment. Supplementary statistics such as number of observations and standard error are also generated. These statistics can aid determinations when an AUID is just above or just below the WQS or where stressor and response variables are not in full agreement.
- **Method detection limits (MDL) for BOD5 data**. For most RES parameters, MDLs will not be an issue during assessments. For example, MDLs for TP (typically <10 µg/L) and chl-*a* (typically <0.5 µg/L) are well below the water quality standards (WQS) and less than values are uncommon. However, BOD5 MDLs may vary among laboratories. MDL for BOD5 data used in rule development was 0.5 mg/L (from MDH), which is well below the WQS. In other laboratories, the MDL may be 2.0 mg/L or higher. These MDLs are at or above the WQS for the North and Central RNRs and in some cases the South RNR. Following are cautions and considerations on the use of BOD5 non-detect data in RES assessment (see also Figure 5). While BOD5 is referred to specifically these considerations would also be applicable to TP and chl-*a* data where high MDLs were used and numerous non-detects are present in the assessment data.
    1. If the BOD5 average is above the WQS and there are no non-detects, then the parameter does not meet the WQS.
    2. If the BOD5 average is below the WQS, regardless of presence of non-detects, the parameter meets the WQS.
    3. If the BOD5 average is above the WQS and non-detects are present, there are several methods that can be used for assessment depending on the dataset. These methods should be followed in sequence.
    4. If the BOD_5 average is above the WQS, but with more than 50% non-detects, the data is considered insufficient information.
    5. If the BOD_5 average is above the WQS and 50% or fewer are non-detects then:
        a. Replace non-detects with “0” and recalculate the mean. If the recalculated mean is still above the standard, the concentration can be considered to exceed the standard. [The occurrence of non-detects in a dataset will increase the mean above the true value. This is because the reported non-detect value is higher than the true value. A simple method to determine if non-detects are potentially biasing the assessment is to use a best-case scenario. This is accomplished by replacing non-detects with “0” values. Since the true value is somewhere between the detection limit and “0”, this recalculated represents the lowest possible mean value.]
        b. If replacing non-detects with “0” results in a recalculated mean that is below the standard, then more sophisticated mean estimation methods are required. If the BOD_5 data are critical to the assessment, advanced non-detect methods such as NADA in “R” may be required to allow for a more accurate estimate of the mean value. If the minimum detection limit for non-detect samples was greater than 0.5 mg/L, the data should not be used for assessment as such data was not used in the analysis for the WQS development.

Figure 5. Flow chart for addressing dataset containing non-detects.



- **pH assessment.** Since pH assessments are based on the existing pH WQS, assessments should be done in accord with the existing methodology (i.e., the variable exceeds the standard if the data show a 10% exceedance of the WQS based on daily minimum and maximum measurements); however, pH data must be collected during the summer index period to be used as a part of RES assessment.
- **Periphyton assessment.** Due to the intensive nature of periphyton data collection it is likely assessment will be based on two sample events over two years. Multiple samples on the same day are averaged. If multiple samples occur on a reach in a given summer, the maximum daily average is used. The standard is exceeded if concentrations exceed 150 mg/m^2 more than one year in ten.
- **Diel DO flux assessment.** Diel DO flux values are calculated based on the difference between the daily maximum DO and the daily minimum DO. These daily flux values are averaged based on the number of days of measurement. Heiskary et al. (2013; Table 6) provides an example of how

data can be assembled for RES assessment purposes. The resulting average diel DO flux measurement is then compared to the WQS to determine if this response variable is met or exceeded.

- **Exceedances of BOD_5 or diel dissolved oxygen flux caused by other factors.** Indirect response measures can be influenced by other factors, which must be considered during the assessment. As with all assessment parameters, each is individually reviewed to determine if the site location was appropriate, if flow conditions and sampling regime were representative (e.g., not biased by flood or drought), and to ensure that there are no quality assurance issues with the data (e.g., data out of hold time, sonde calibration issues). When reviewing BOD_5 data, the proximity to permitted facilities must be taken into account as data included in the assessment may be within the mixing zone of the facilities discharge. These locations should be reviewed to determine if the discharge is biasing the values. For diel DO flux, flow conditions during deployment should be examined to determine if flow conditions were not typical and impacted diel DO flux measurement.
- **Clear evidence of WQS exceedance.** AUIDs exceed the RES if the causative variable (TP) exceeds the standard and one or more of the response variables (chl-*a*, BOD_5, diel DO flux, pH or periphyton) also exceed the standard. Such AUIDs are impaired and the AUID will be included on Minnesota's 303(d) list. Not all response variables need to be present or in agreement for an exceedance to be determined.
- **Clear evidence of meeting the WQS.** An AUID is meeting the RES if total phosphorus is meeting the standard. A determination of full support of the RES does not require response data to be present. However, if response variable data are present and assessable, a determination of full support requires that the response variables also meet the applicable standard. An AUID can also be considered fully supporting if total phosphorus exceeds the threshold and all response variables can be assessed and they meet their respective standard.
- **Insufficient information to assess.** A determination of insufficient information will be assigned when:
  1. Insufficient data are present.
     a. Insufficient total phosphorus data available.
     b. Sufficient total phosphorus data are available and indicates exceedance of the standard, but no response variable data are present.
  2. Sufficient data for assessment exists, but there is a lack of confidence in the data (e.g., inappropriate laboratory methods, atypical flow conditions, inappropriate sample location).
- **Average concentrations near the standard.** AUIDs where TP or response variable(s) are slightly above or slightly below the WQS require closer scrutiny of the data. A high standard error (SE), indicative of high variability in measurements, suggests the raw data should be reviewed to determine the frequency of elevated values. If TP ± SE is just above the WQS but response WQS are met, the reach is deemed supporting the WQS. If TP ± SE is just above the WQS and mean chl-*a,* BOD_5, diel flux or pH exceeds the WQS, the reach is deemed not supporting aquatic life use due to eutrophication. If the data are not representative, such as poor site placement (i.e., lake outlet, in mixing zone of permitted facility), data skewed by drought- or flood-biased samples, etc. the reach may be considered insufficient information to assess. If flow data are available, this may help place results in perspective. For example, if summer-mean chl-*a* is equal to the response WQS but collections were made only during high flow summers, it is likely chl-*a* would exceed in summers with lower flow and it may be reasonable to recommend listing the AUID if TP exceeds as well. A recommendation of not listing may be reasonable if collections were made only during low flow summers.

- **Effect of impoundment (≥14-day residence time) upstream or within the AUID.** An impoundment immediately upstream or in the AUID may promote excessive algal growth even when TP meets the river eutrophication WQS. In instances like this, a decision may be needed as to whether the lake or river eutrophication WQS is most appropriate to address this situation. In cases where the upstream impoundment has been deemed a reservoir and was assessed as impaired (based on the lake eutrophication standard (LES)), the "assessment status" of the river AUID may not affect the TMDL since the TMDL for the impoundment would likely address the river eutrophication issue.
- **Effect of impoundment (<14-day residence time) upstream or within the AUID.** Very small or short residence time impoundments or wetland complexes on the mainstem of a river (residence time < 14 days at 122-day one in 10-year low flow) represent a special case and there is a need to determine the status of data collected from reaches affected by these impoundments or wetlands in terms of 1) whether or not the data are assessable, 2) which if any standard is appropriate, and 3) how it may influence a downstream portion of the AUID. To determine if a river reach is impounded a review of dam location (DNR GIS layer), river morphology (aerial photos, site visits), water velocity, etc. will be used. The RES and LES standards were developed using data from un-impounded river stations and lakes that met the 14-day residence time threshold, respectively. These datasets did not include naturally or artificially impounded river reaches so the applicability of the either standard needs to be determined on a case-by-case basis. In most instances, best professional judgment will be used and documented to discern which standard is appropriate for the AUID in question. However, in some cases there will not be sufficient supporting information to determine an appropriate standard and data from the impounded section will need to be flagged as supporting information only. When an AUID includes data from both an impounded and un-impounded reaches, the data from the un-impounded reach may still be assessable against the RES standard.
- **Biased data.** As a part of the data review for assessments, RES datasets should be examined to identify possible biases resulting from irregular timing of sampling (e.g., samples weighted toward part of the year or to high flow events). If the data are not representative of the index period, a time-weighted average can be applied to correct this bias [note this procedure will only be needed when the bias is likely to have a significant impact on the assessment]. In addition to removing within-year temporal biases, the time-weighted average will also weight data from each year equally to reduce weighting toward years with larger sample sizes. However, caution should be used with data from years with few sample events (<4) or with data from only part of the year (e.g., only August samples). Years with only a single sample should be removed from the time-weighted calculation as the temporal weighting cannot be calculated for these years and the single sample would be given too much weight. Years with only two-three sample events should be scrutinized to determine how well the limited sample size reflects average annual conditions. These data may be removed or retained depending on this evaluation. Any data that are removed may still be useful as supporting information.

  A time-weighted average can be calculated using the following equation.

$$TWA = \frac{\sum_1^n c_i * t_i}{\sum_1^n t_i}$$

  where c_i = concentration for the i^{th} sample
  t_i = time window for the i^{th} sample

- **Site-specific standards option.** Sometimes it is more appropriate and information is available to derive standards based on information specific to an AUID. Site-specific standards require public comment and must be sent to EPA for approval. Additional data collection work may be required to develop and adopt a proposed site-specific WQS. Once approved, the site-specific standard becomes the basis for assessing the condition of the AUID.
- **Use of data near continuous discharging facilities.** BOD_5 and DO flux data from within five miles of a continuously discharging wastewater treatment facility (WWTF) are generally not valid for assessing RES. The intent of these response variables is to identify the presence of eutrophication (i.e., undesirable levels of sestonic or suspended algae, benthic or attached algae, or excessive rooted vegetation). Some river monitoring sites are too close to WWTF outfalls and are biased by dying microbial matter and not algae or rooted vegetation. A 2010 MPCA paper analyzed data and determined that in most instances, data from within five miles downstream of a facility may be impacted by the effluent. As a result, it would not be appropriate to use these values in a RES assessment.

Mississippi navigational pool assessments

Navigational pool eutrophication assessments on the Mississippi River should be consistent with other 303(d) assessments; whereby the most recent 10 years of data would be used in the assessment. This should minimize the effect of any extreme high or low flow year and allow for a more comprehensive assessment of each assessment reach.

- Assessments will be based on monitoring data collected in the thalweg of the pools just upstream of the dam that forms the pool. The monitoring sites should be consistent with long-term monitoring sites employed by the Metropolitan Council (MCES) and USGS's Long Term Resource Monitoring Program (LTRMP) (see Table 14). The pool is designated as impaired if TP and chl-*a* exceed the WQS as noted in Table 4 of Heiskary and Wasley (2012).

Table 14. Station data used for Mississippi River pools assessments

Pool	AUID	Stations used for standard development and assessment
Pool 1	07010206-814	MCES 847.7, EQuIS S004-276
Pool 2	07010206-814	MCES 815.6, EQuIS S000-068
Pool 3	07040001-531	LTRMP M796.9, MCES 796.9, EQuIS S005-179, S000-132
Pool 4/Lake Pepin	25-0001-00	LTRMP M766.0I, 771.2P, 775.6Q, 781.2O
Pool 5	07040003-627	LTRMP M738, EQuIS S000-287
Pool 6	07040003-627	EQuIS S000-095
Pool 7	07040006-515	LTRMP M701.1
Pool 8	07060001-509	LTRMP M679.5, EQuIS S000-094

- **Lake Pepin assessments**. Lake Pepin assessments will be based on fixed site monitoring data and incorporate the most recent 10 years of data. This data is collected at two sites in the upper segment and two sites in the lower segment of the lake and correspond to long-term sites that have been used by LTRMP and MPCA. Data from these four sites were the primary basis for listing Lake Pepin as impaired and supported much of the model development and testing. Data from all four sites are averaged for the assessment. Site maps and further description are found in Heiskary and Wasley (2011).

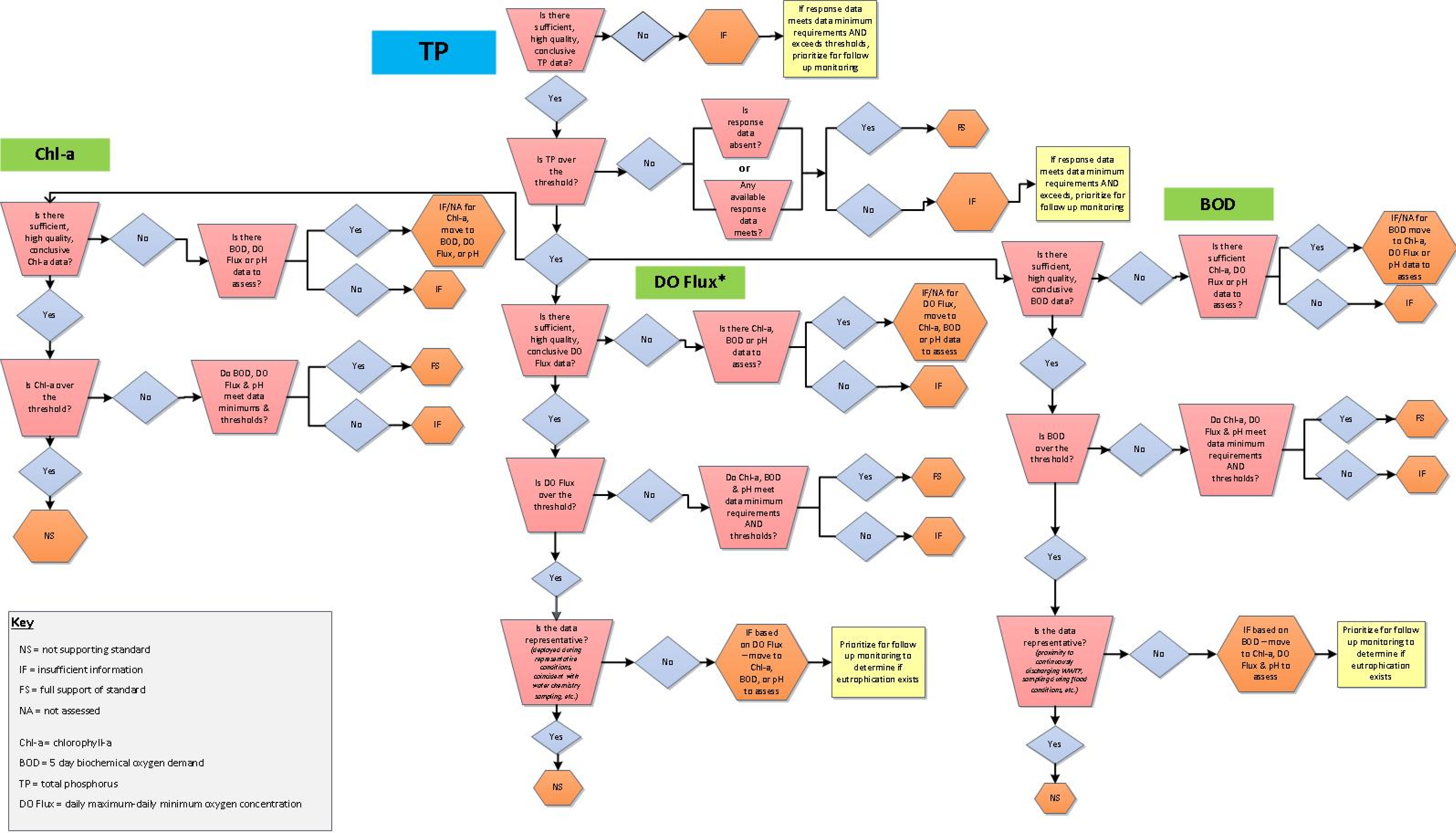
Special assessment situations related to RNR assignment

When assessments are made or new AUIDs are established, there may be a need to assign new RNRs or to change an RNR designation because of new information that is gathered in the assessment process. This may occur as a part of the professional judgment group review, as a result of public comment, or in the course of TMDL development. In some instances, this may require some correction in RNR designation, while in others it may require development of a site-specific standard.

Some stream reaches may require site-specific standards within the context of the RNRs (Figure 4). These situations most often occur when two similar order (sized) rivers from two different RNRs join prior to discharging to a major downstream, higher order river. For example, in adoption of the river eutrophication standards Exhibit EU-5 notes: *"In a few instances where two HUC-8s meet prior to entering the major mainstem river (e.g. North Fork and South Fork Crow Rivers) "blended" or site-specific standards are recommended and these reaches are noted on the RNR map."* Where and when such sites are identified in the future, the site-specific WQS for the causative variable (TP) is likely to be based on the midpoint between the values from the two contributing RNRs. The site-specific WQSs for the response variables will be based on the midpoint between the WQS in Table 11 of Heiskary et al. 2013. This approach and values as noted in Table 11 of Heiskary et al. 2013 should be applicable in other instances where this may occur.

Table 15. Minnesota's site-specific river eutrophication standards

	Causative	Response (stress)		
Region or river	**Total phosphorus µg/L**	**Chlorophyll-*a* (seston) µg/L**	**Diel dissolved oxygen flux mg/L**	**Biological oxygen demand mg/L**
Mississippi River Navigational Pool 1	100	35		
Mississippi River Navigational Pool 2	125	35		
Mississippi River Navigational Pool 3	100	35		
Lake Pepin (Mississippi River Navigational Pool 4)	100	28		
Mississippi River Navigational Pools 5 to 8	100	35		
Crow Wing River from Long Prairie River to the Mouth of the Crow Wing River	75	13	3.5	1.7
Crow River from the confluence of the North Fork and South Fork of the Crow River to the mouth of the Crow River	125	27	4.0	2.5



*pH follows the same decision tree as DO Flux

Calibration of the biological condition gradient in Minnesota streams: a quantitative expert-based decision system

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Abstract: The Biological Condition Gradient (BCG) is a conceptual model that describes changes in aquatic communities with increasing levels of anthropogenic stress. The gradient represented by the BCG has been divided into 6 levels of condition that biologists consider readily discernible in most areas of North America. We developed quantitative BCG models for 7 warm-water stream types in Minnesota for both fish and macroinvertebrates. Panels of aquatic biologists calibrated the general BCG model to Minnesota streams by assigning test samples (271 macroinvertebrate and 288 fish samples) to BCG Levels 1 to 6. From the panelists' descriptions of their criteria for assigning sites to levels, a set of quantitative operational rules was developed for performing the same task. We developed a decision model based on fuzzy-set theory to account for discontinuities and to identify when BCG assignments might be intermediate between adjacent levels. This model captures the consensus professional judgment of the panel and uses panel-derived rules. Decisions based on the quantitative model for macroinvertebrates exactly matched 77% of the panel decisions, 89% within ½ BCG level, and 100% within 1 BCG level. Decisions based on the quantitative fish model exactly matched 70% of the panel decisions, 86% within ½ BCG level, and 99% within 1 BCG level. The BCG provides a tool to interpret aquatic biological condition along a gradient of naturalness and is consistent across stream types and political boundaries. It includes documentation of baselines to prevent inadvertent shifting, and the BCG logic rules are transparent, a desirable property for communicating condition, management goals, and water-quality criteria.
Key words: Biological Condition Gradient, decision model, fuzzy logic, expert system, Minnesota, benthic macroinvertebrates, fish, water quality management, streams

In many nations, policies developed to protect and maintain water quality include the concepts of biological and ecological quality, which are assessed on the basis of the ecological structure and function of living aquatic communities. The US Clean Water Act of 1972 (CWA) has the long-term objective of restoration and protection of chemical, physical, and biological integrity (US Code title 33, §1251 [a]; USEPA 2011). In the European Union (EU), the Water Framework Directive (WFD) has the similar objective of restoration and maintenance of 'good' or better ecological quality (e.g., Hering et al. 2010, EU Commission 2015). Both the US Environmental Protection Agency (EPA) and the EU have made efforts to define what was meant by 'biological integrity' (USA) and 'high', 'good', 'fair', 'poor', and 'bad' condition (EU). In the USA, biological integrity has come to mean "The ability of an aquatic ecosystem to support and maintain a balanced, integrated and adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats within a region" (Frey 1977, Karr and Dudley 1981). In the EU, high ecological quality is defined as the ecological condition occurring under "no or very low human pressure" and is accepted as the reference condition (EU Commission 2015). Good through bad condition are

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defined as successively increasing deviation from high or reference status (Hering et al. 2010). Both systems use natural condition with no or minimal human influence as a benchmark.

To meet the goals of the CWA and WFD, ecologically consistent interpretations of biological condition are needed to allow definition of thresholds of condition for assessment, restoration, and management. The definitions must be specific, well-defined, and must allow for waters of different natural quality and different desired uses. In the USA, the EPA developed a conceptual model—the Biological Condition Gradient (BCG)—that describes ecological changes from pristine to severely degraded that occur in flowing waters with increased anthropogenic degradation (Davies and Jackson 2006). The BCG was designed to provide a way to map different indicators on a common scale of biological condition to facilitate comparisons among programs and across jurisdictional boundaries. The original BCG is a conceptual, narrative model that describes how biological attributes of aquatic ecosystems change along a gradient of increasing anthropogenic stress (Fig. 1) and provides a framework for understanding current conditions relative to natural, undisturbed conditions (Davies and Jackson 2006, USEPA 2016).

US states, EU member states, and academics and environmental agencies worldwide have developed technical approaches and indexes to assess the biological condition of water bodies. In recent years, most approaches have been variations on the multimetric Index of Biotic Integrity (IBI; Karr et al. 1986, Whittier et al. 2007, Pont et al. 2009) or multivariate interpolations of reference-site species composition (River Invertebrate Prediction and Classification System; RIVPACS; e.g., Hawkins et al. 2000, Simpson and Norris 2000, Wright 2000). These indexes rely on empirical, present-day reference conditions quantified from existing reference sites to anchor their measurement systems. They require 'minimally disturbed' reference sites that are representative of biological integrity (Stoddard et al. 2006). However, in practice, most reference site data sets consist of 'least-disturbed' sites, which are the best remaining sites. The distinction between minimally disturbed and least-disturbed is important: minimally disturbed denotes fully natural biological conditions indistinguishable from pre-industrial or pre-European settlement, whereas least-disturbed denotes an upper quantile of contemporary conditions (Stoddard et al. 2006). Most indexes are built from a statistically adequate sample of least-disturbed (best available) reference sites, so that 1 or 2 minimally disturbed (near-pristine) sites in a reference data set may be treated as statistical outliers and may have little influence on index scoring. In the situation where no reference sites meet minimally disturbed criteria, the best score of this index would be similar to the moderately disturbed reference sites and could be substantially degraded from the natural condition. This situation is an example of the 'shifting baseline syndrome', such that the ideal reference or condition changes over generations as memory of previous baselines is lost (e.g., Pauly 1995, Dayton et al. 1998).

Part of the BCG process is to build a description of a fixed baseline based on either minimally disturbed conditions (Stoddard et al. 2006) or a fixed, agreed-upon point in time. The initial description is based on professional judgment, but as the BCG approach becomes accepted, the professional judgment should be replaced or enhanced

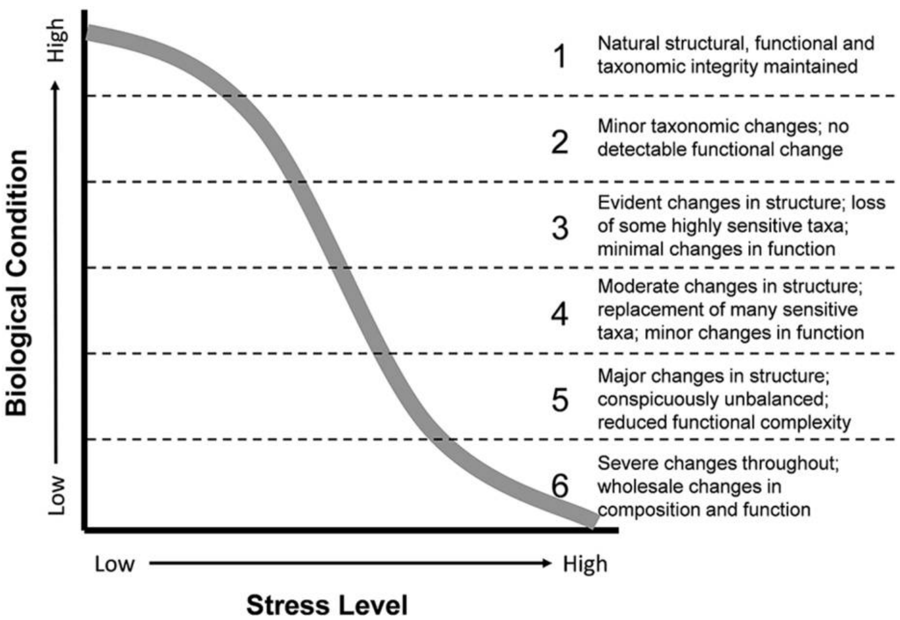

Figure 1. Graphic representation of the Biological Condition Gradient conceptual model (modified from Davies and Jackson 2006 with permission from John Wiley and Sons).

with documented information: historical descriptions, paleo investigations, museum records, and information from documented minimally disturbed sites. The description of minimally disturbed is necessarily incomplete, but its documentation is a defense against future inadvertent baseline shifts. Careful use of the BCG would identify a natural or historic baseline that could be used to guard against shifting baseline syndrome. For regions or situations where all information on natural baseline is irretrievably lost, the BCG could assist in identifying an 'Anthropocene baseline' for restoration and management (Kopf et al. 2015).

The quantitative BCG development was published by the USEPA (2016) based on case studies from the preceding decade. The methods have matured and experience gained has shown that a quantitative BCG has several desirable properties for use in water-quality management:

1. *Universal interpretive scale based on biologically meaningful changes* The original intent of the BCG was to create a scale with uniform interpretation across political and administrative jurisdictions (Davies and Jackson 2006). This intent was in response to the risk that use of different biological indexes and thresholds might result in contrary interpretations among states, wherein one state might call a cross-border stream impaired, but a neighboring state might not.
2. *Documented defense against shifting baselines* BCG values and thresholds are designed to defend against shifting baselines by including a description of undisturbed conditions. Any index or assessment method can include a documented baseline, but many indexes have been built empirically with data from 'least-stressed' reference sites (Stoddard et al. 2006). The BCG is independent of sometimes arbitrary percentiles of empirical reference populations. In the USA, management criteria consisting of the 50^{th}, 25^{th}, 10^{th}, 5^{th}, and 0^{th} percentiles of reference distributions have all been proposed by states and advocacy groups.
3. *A transparent decision system with stated rules* The quantitative BCG method consists of documented decision rules and, therefore, is transparent. Rules can be changed, but changes are conscious and deliberate and cannot result from additions or deletions in a database. The decision system provides a bridge between ecological science and value-based management. BCG levels can be adopted directly as management goals, restoration goals, or regulatory (protective) criteria.
4. *Flexibility* A quantitative BCG model can be used as a stand-alone assessment index or cross-walked to other existing indexes to provide ecological interpretation and identify management thresholds (Bouchard et al. 2016).

Here, we explain the calibration of a quantitative assessment model in the framework of the BCG. We use as an example the development of the model for warm-water streams and rivers of the state of Minnesota, USA, for benthic macroinvertebrate and fish assemblages (original report: Gerritsen et al. 2013).

METHODS

BCG primer

Biologists from across the USA developed the BCG conceptual model and agreed that a similar sequence of biological alterations occurs in streams in response to increasing stress, even in different geographic regions (Davies and Jackson 2006). The BCG is divided into 6 levels of biological condition along the stressor–response curve. Levels range from observable biological conditions found at no or low levels of stress (Level 1) to those found at the highest levels of stress (Level 6) (Fig. 1, Table 1). The 6 levels of the BCG are convergent with the 5 ecological status conditions defined in the EU WFD. The BCG levels were described in greater detail by Davies and Jackson (2006).

The BCG uses 10 attributes of aquatic ecosystems that change in response to increasing levels of stress along the gradient to describe the 6 levels (Table 2). The attributes include aspects of community structure, organism condition, ecosystem function, spatial and temporal attributes of stream size, and connectivity and are used as indicators of condition. The BCG was developed originally based on forested streams of eastern North America as examples (Davies and Jackson 2006), but the model has been applied to other regions and water bodies by calibrating it to local conditions on the basis of specific expertise and local data. Several US states, tribes, and territories are calibrating BCG-based indexes based on the first 7 attributes that characterize the biotic community, primarily tolerance to stressors, presence/absence of native and nonnative species, and organism condition (Table 2). BCG models have been developed for streams, lakes, estuaries, and coral reefs and biological assemblages including fish, benthic macroinvertebrates, and diatoms (summarized by USEPA 2016; Gerritsen and Leppo 2005, Stamp and Gerritsen 2012, Hausmann et al. 2016, Santavy et al., in press).

Approach

Our approach for BCG model development is based on professional judgment and development of consensus. Professional consensus has a long pedigree in the medical field, including the National Institutes of Health (NIH) Consensus Development Conferences to recommend best practices for diagnosis and treatment of diseases (http://consensus.nih.gov/). The NIH consensus meetings were a "hybrid of . . . judicial decision-making, scientific conferences and the town hall meeting" (Nair et al. 2011). Other researchers, institutes, and countries also develop medical consensus statements using NIH methods (Nair et al. 2011).

Table 1. Descriptions of Biological Condition Gradient levels (modified from Davies and Jackson 2006).

BCG level	Description
Level 1: Natural or native condition	Native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability. Level 1 represents biological conditions as they existed (or may still exist) in the absence of measurable effects of stressors.
Level 2: Minimal changes in structure of the biotic community and minimal changes in ecosystem function	Virtually all native taxa are maintained with some changes in biomass or abundance; ecosystem functions are fully maintained within the range of natural variability. Level 2 represents the earliest changes in densities, species composition, and biomass that occur as a result of slight elevation in stressors (such as increased temperature regime or nutrient enrichment).
Level 3: Evident changes in structure of the biotic community and minimal changes in ecosystem function	Evident changes in structure caused by loss of some highly sensitive native taxa; shifts in relative abundance of taxa but sensitive-to-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system. Level 3 represents readily observable changes that, e.g., can occur in response to organic enrichment or increased temperature.
Level 4: Moderate changes in structure of the biotic community with minimal changes in ecosystem function	Moderate changes in structure caused by replacement of some intermediate-sensitive taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes.
Level 5: Major changes in structure of the biotic community and moderate changes in ecosystem function.	Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from those expected; organism condition shows signs of physiological stress; ecosystem function shows reduced complexity and redundancy; increased build-up or export of unused materials. Changes in ecosystem function (as indicated by marked changes in foodweb structure and guilds) are critical in distinguishing between Levels 4 and 5.
Level 6: Severe changes in structure of the biotic community and major loss of ecosystem function	Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered. Level 6 systems are taxonomically depauperate (low diversity or reduced number of organisms) compared to the other levels.

Experts define BCG levels in the context of the conceptual model (Davies and Jackson 2006). They determine the attributes and the changes in those attributes that characterize distinct BCG levels and signal shifts to a different level (Tables 1, 2). The BCG consensus approach asks the experts to make judgments on the biological significance of changes in the attributes. Thus, a fundamental assumption of this approach is that consensus professional judgment is the best current estimate of biological condition. The outcome of the process is a multiple-attribute decision model that mimics the consensus decisions based on a set of quantitative rules. The logic train of the decision model and the experts' documented reasoning create a transparent decision system for review, modification, and water-quality management.

Index calibration begins with the assembly and analysis of biological monitoring data and identification of stress–response relationships for individual taxa. During one or more calibration workshops, experts familiar with local conditions and biota use the data to develop narrative decision rules for assigning sites to a BCG level. Panelists assign relevant taxa to BCG attributes (Table 2). Next, they examine biological data from selected sites, describe the native aquatic assemblages under natural conditions, and assign the samples to Levels 1 to 6 of the BCG. The intent is to achieve consensus and to identify rules that experts use to make their assignments. Experts' opinions are elicited and documented to assist in quantitative rule development.

Over the long term, reconvening the same group of experts for every new sample is impractical. Thus, use of a quantitative BCG in routine monitoring and assessment requires a way to automate the consensus expert judgment. The decision criteria are codified into a quantitative decision model, which is a transparent, formal, and testable method for documenting and validating expert knowledge.

For over a decade, the Minnesota Pollution Control Agency (MPCA) has been using fish and benthic macroinvertebrate assemblage data to assess water resource quality. Until recently, biological indexes in Minnesota were developed for individual drainage basins (e.g., Niemela et al. 1999). The MPCA used data from 2285 fish and 1502 macroinvertebrate samples to develop statewide fish and macroinvertebrate IBIs following the approach published by Whittier et al. (2007). Descriptions of these IBIs can be found in MPCA (2014b, c). The BCG calibration we describe here relies heavily on the knowledge and experience gained from Minnesota's IBI developments, and addresses MPCA's ob-

Table 2. Attributes used to characterize the Biological Condition Gradient (BCG) (modified from Davies and Jackson 2006).

Attribute	Description
	Attributes I–V: Native structure and composition
I. Historically documented, sensitive, long-lived, or regionally endemic taxa	Taxa known to have been supported according to historical, museum, or archeological records, or taxa with restricted distribution (occurring only in a locale as opposed to a region), often because of unique life-history requirements (e.g., sturgeon, American Eel, pupfish, unionid mussel species)
II. Highly sensitive (typically uncommon) taxa	Taxa that are highly sensitive to pollution or anthropogenic disturbance; tend to occur in low numbers, and many are specialists for habitats and food type; the first to disappear with disturbance or pollution (e.g., most stoneflies, Brook Trout [in the eastern USA], Brook Lamprey)
III. Intermediate sensitive and common taxa	Common taxa that are ubiquitous and abundant in relatively undisturbed conditions but are sensitive to anthropogenic disturbance/pollution; have a broader range of tolerance than attribute II taxa and can be found at reduced density and richness in moderately disturbed sites (e.g., many mayflies, many darter fish species)
IV. Taxa of intermediate tolerance	Ubiquitous and common taxa that can be found under almost any conditions, from undisturbed to highly stressed sites; broadly tolerant but often decline under extreme conditions (e.g., filter-feeding caddisflies, many midges, many minnow species)
V. Highly tolerant taxa	Taxa that typically are uncommon and of low abundance in undisturbed conditions but increase in abundance in disturbed sites; opportunistic species able to exploit resources in disturbed sites; the last survivors (e.g., tubificid worms, Black Bullhead)
VI. Nonnative or intentionally introduced species	Any species not native to the ecosystem (e.g., Asiatic clam, Zebra Mussel, carp, European Brown Trout); in addition, many fish native to one part of North America introduced elsewhere
VII. Organism condition	Anomalies of the organisms; indicators of individual health (e.g., deformities, lesions, tumors)
VIII. Ecosystem function	Processes performed by ecosystems, including primary and secondary production, respiration, nutrient cycling, decomposition, their proportion/dominance, and what components of the system carry the dominant functions (e.g., shift of lakes and estuaries to phytoplankton production and microbial decomposition under disturbance and eutrophication)
IX. Spatial and temporal extent of detrimental effects	The spatial and temporal extent of cumulative adverse effects of stressors (e.g., groundwater pumping in Kansas led to change in fish composition from fluvial-dependent to sunfish)
X. Ecosystem connectivity	Access or linkage (in space/time) to materials, locations, and conditions required for maintenance of interacting populations of aquatic life; the opposite of fragmentation (e.g., levees restrict connections between flowing water and floodplain nutrient sinks, dams impede fish migration, spawning)

jective to develop statewide biological criteria for streams within Minnesota.

Aquatic biologists familiar with Minnesota streams met as a work group to develop the ecological attributes and rules for assigning sites to levels. Their expertise included aquatic ecology, benthic macroinvertebrate sampling and monitoring, water quality, and fisheries biology. We summarize here the results of BCG calibration for warm-water streams in Minnesota (Gerritsen et al. 2013). A 2nd multistate and multi-tribal effort to develop a BCG calibration for cold water streams of the Upper Midwest was reported by Gerritsen and Stamp (2013).

Data

When the models were developed, the MPCA had collected >3800 fish and >2800 macroinvertebrate samples from warm-water streams (1996–2011). Minnesota's biological assessment program was assessed in 2015 (USEPA 2013) and was deemed sufficient to support development and implementation of biological monitoring tools (MBI 2015).

A fish sampling reach is defined as 35× mean stream width. This length is sufficient to capture a representative and repeatable sample of the fish assemblage in a stream segment (Lyons 1992, MPCA 2014d). Sampling is conducted during daylight hours in the summer index period (mid-June–mid-September). Streams are sampled during or near base flow because floods or droughts can affect fish assemblage structure and sampling efficiency. All habitat types within the sampling reach are sampled in approximate proportion to their occurrence to capture fish ≥25 mm in total length. Four electrofishing methods are used: backpack electrofisher in small headwater streams; towed stream electrofisher in larger wadeable streams; mini-boom electrofisher (2-person jon boat) in small, nonwadeable streams;

and a boat-mounted boom electrofisher in large streams and rivers. For detailed fish sampling methods see MPCA (MPCA 2014d). Fish sampling is repeated at 10% of the sample reaches during the index period to estimate measurement error.

A multihabitat method is used to obtain a representative sample of the macroinvertebrate assemblage of a reach. Habitats sampled include hard bottom (riffle/cobble/boulder), aquatic macrophytes (submerged/emergent vegetation), undercut banks (undercut banks/overhanging vegetation), snags (snags/rootwads), and leaf packs. Twenty D-frame dipnet (500-μm mesh) sweeps are divided equally among the dominant, productive habitats present in the reach. Each sweep covers ~0.09 m^2 of substrate for a total area sampled of ~1.8 m^2. Collections are randomly subsampled to a target subsample of 300 individuals and identified to genus. Macroinvertebrate collection standard operating procedures (SOPs) were described fully by the MPCA (MPCA 2014e). Macroinvertebrate sampling is repeated at 10% of the sample reaches on the same day to estimate measurement error.

Measurement error (sample variability) was not estimated as part of this project, but Minnesota's sampling and analysis methods are comparable to those used by EPA in national aquatic surveys (e.g., Stoddard et al. 2008). Other studies of similar methods have shown variability of indexes to be low and consistent for repeated samples within and among years (e.g., Hose et al. 2004, Barbour and Gerritsen 2006, Huttunen et al. 2012).

Classification

Classification of aquatic habitats is necessary to account for natural variability so that the experts can place a stream in context of its setting. Panelists involved in some early attempts to develop a quantitative BCG struggled in the absence of a classification scheme understood by the panel and appropriate for the data set (USEPA 2016). Most panels have preferred a primarily typological classification (e.g., ecoregions), but continuous classifiers, such as catchment area, stream gradient, and elevation, have been used successfully.

The MPCA developed a classification system for natural stream communities to support the development of typological IBI models (MPCA 2014b, c). The stream types were based on distributions of species among classification variables that are not influenced by anthropogenic effects. The classification system for warm-water streams was developed with the same data set used to develop the IBIs and consisted >2200 fish and 1500 macroinvertebrate samples collected from 1996 through 2008. Biological communities and predictive variables were identified with the aid of several tools including: hierarchical cluster analysis, nonmetric multidimensional scaling, and Mean Similarity Analysis (Van Sickle 1998, Van Sickle and Hughes 2000). This process resulted in 7 warm-water stream types each for the fish and the benthic macroinvertebrate communities based on: 1) ecoregion, 2) sampling method, 3) drainage area, and 4) stream gradient (Table 3). Fish and macroinvertebrate stream types follow a similar regional pattern, but they do not match. For example, invertebrate high-gradient and low-gradient habitats may occur in both wadeable and headwater streams as defined for fish sampling. Geographic delineations included northern or southern Minnesota and forest or prairie. The remaining classes were defined by sampling method (e.g., high-gradient vs low-gradient for macroinvertebrates).

Preliminary analysis: stress–response and BCG attributes

The MPCA developed a disturbance index called the Human Disturbance Score (HDS) based on the degree of human activity in the upstream watershed and at the reach level for biological monitoring sites (Bouchard et al. 2016, MPCA 2016). The HDS includes 8 primary metrics, which consist of measures of watershed land use, stream alteration, riparian condition, and known permitted discharges.

Table 3. Final Minnesota Pollution Control Agency (MPCA) classifications of warm-water stream types for fish and macroinvertebrates, and number of samples with valid data in each. The 2 river classes correspond between fish and macroinvertebrates, but the wadeable stream classes do not correspond.

Fish stream type		Macroinvertebrate stream type	
Name	*N*	Name	*N*
Northern rivers	358	Northern forest rivers	125
Southern rivers	525	Prairie and southern forest rivers	155
Northern streams	523	Northern forest streams, high-gradient	271
Northern headwaters	706	Northern forest streams, low-gradient	425
Southern streams	665	Southern streams, high-gradient	445
Southern headwaters	638	Southern forest streams, low-gradient	396
Low-gradient streams	313	Prairie streams, low-gradient	617

HDS scores can range from 1 (heavily altered watersheds) to 81 (nearly pristine watersheds).

Stress-response models

BCG composition attributes II through V (Table 2) are familiar tolerance designations (e.g., Merritt et al. 2008) applied in many IBI and multimetric indexes. Published tolerance values are often 'received wisdom' originally estimated from different regions (Carlisle et al. 2007), so we augmented the published values with analysis of the MPCA data to estimate tolerances from the local data. We used general linear models (GLMs) to estimate the probability of observing a particular taxon across the HDS score. The optimum of the model (maximum probability) yielded the tolerance value. We plotted the capture probabilities over the range of the disturbance gradient (Figs 2–5).

Assign taxa to attributes

Assignments of taxa to attributes relied on a combination of the empirical data analysis (Figs 2, 3A, B, 4A, B, 5A, B), published values, and professional experience of the expert panels (Tables 4, 5). HDS is not a perfect measurement of stressors in a stream reach because it is a general predictor of disturbance. It provided an a priori general stressor gradient that is associated with taxon abundance and probability of occurrence to assist the panel in assigning the BCG attributes. The use of empirical data, published tolerances, ecological theory, and professional experience minimizes the effect of noise in the HDS during BCG development.

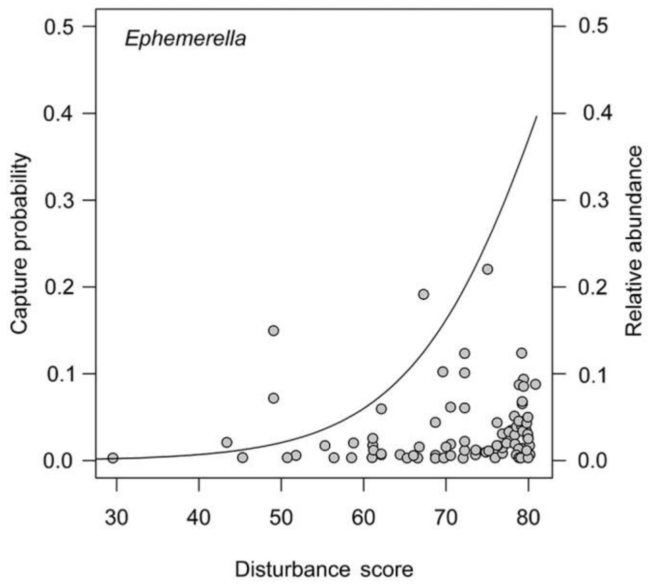

Figure 2. Disturbance score and *Ephemerella* occurrence in stream samples. Circles show observations and relative abundance of *Ephemerella* (right axis); curve shows probability of occurrence (left axis; maximum likelihood). *Ephemerella* was assigned to Biological Condition Gradient (BCG) attribute II (highly sensitive taxa), as shown by its high abundance and high probability of occurrence in minimally disturbed sites (disturbance score 81). See Table 2 for BCG attributes.

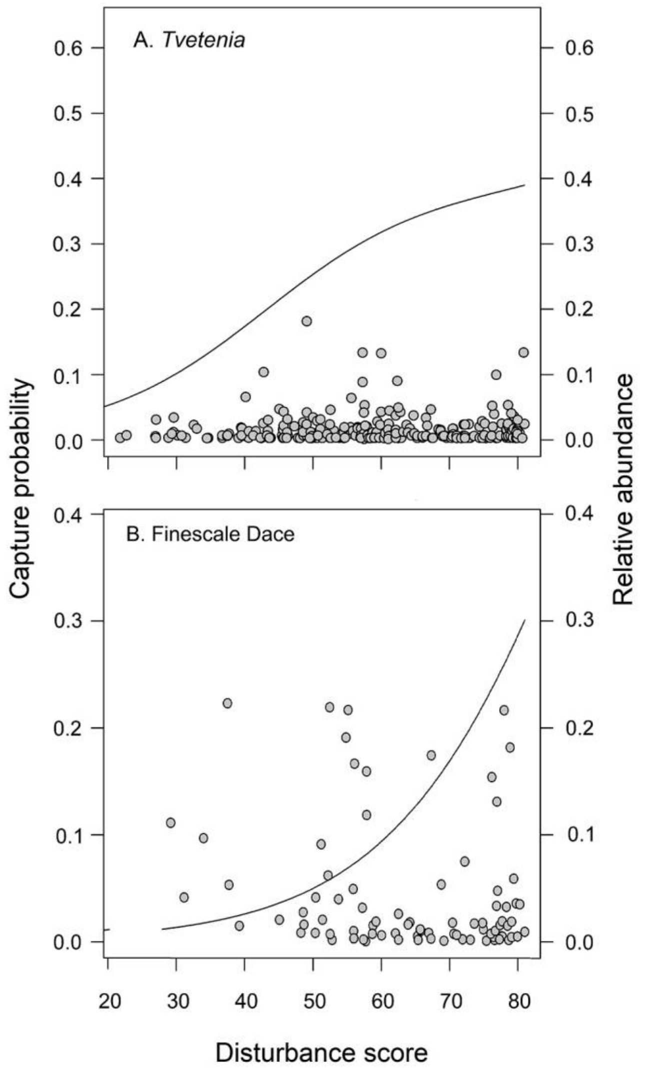

Figure 3. Examples of Biological Condition Gradient (BCG) attribute III taxa *Tvetenia* (A) and Finescale Dace (B). These species occur throughout the disturbance gradient, but with higher probability in better sites. Final attribute assignment was based on these plots and on professional judgment of the panel. See Table 2 for BCG attributes.

For taxa with a sufficient sample, the capture probabilities and, to a lesser extent, the observed abundances followed the expectations given by the attribute descriptions (Table 2, Figs 2, 3A, B, 4A, B, 5A, B). In cases of disagreement, the panel relied on consensus professional opinion unless contradicted by an overwhelming response in the data analysis.

The fish panel identified 2 additional subclasses of the attributes 'tolerant species' and 'nonnative species'. They identified highly tolerant native species (attribute Va) as the last survivors in a degraded stream and divided the

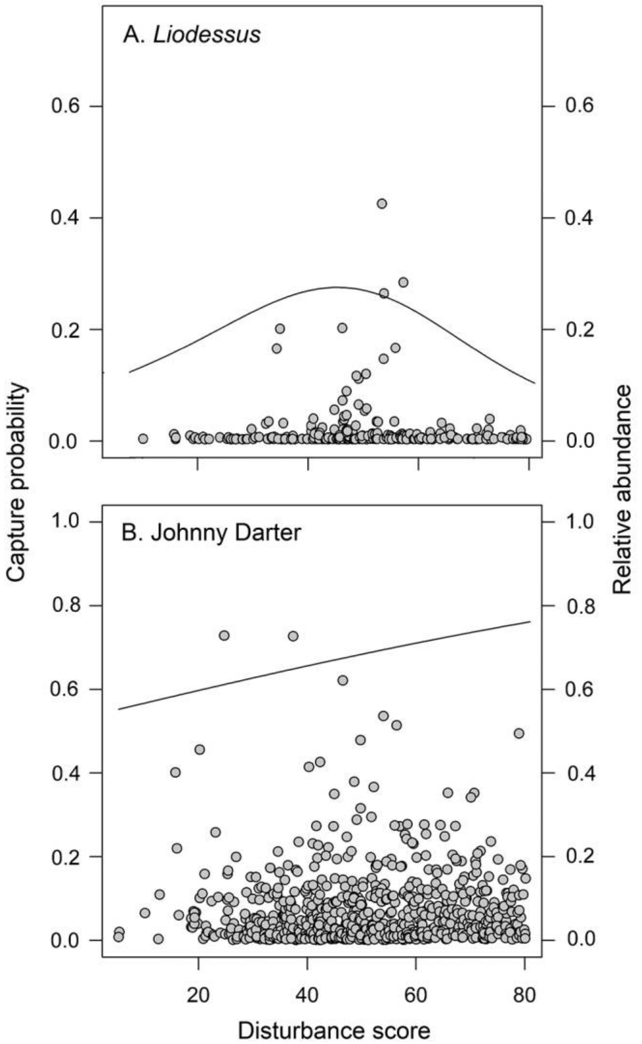

Figure 4. Examples of intermediate tolerant, Biological Condition Gradient (BCG) attribute IV taxa *Liodessus* (A) and Johnny Darter (B). These species occur throughout the disturbance gradient with roughly equal probability throughout or with a peak in the middle of the disturbance range. See Table 2 for BCG attributes.

nonnative group into sensitive nonnative species (attribute VI, e.g., nonnative salmonids) and tolerant nonnatives (attribute VIa; e.g., Common Carp, Ruffe; Table 5).

Assign sites to BCG levels

The panels examined data from selected monitoring sites and assigned the sites to levels of the BCG based on the taxa present in the sample and the generic descriptions of BCG levels (Table 1). The data included lists of taxa and abundances, BCG attribute groups assigned to the taxa, summary metrics, and limited site information, such as stream type and ecoregion, sampling method, and substrate. Stream location, water quality, and MPCA's disturbance score were not revealed to panel members because doing so might have biased assignments. Panel members discussed the species composition, what they expected to see for each level of the BCG, and then assigned samples to BCG levels. The work groups examined macroinvertebrate data from 271 samples (7 stream types), and fish data from 288 samples (7 stream types).

Quantitative description

In the discussions of BCG assignments, facilitators elicited panelist's reasoning for their decision; e.g., "I expect to see more stonefly taxa in a BCG Level-2 site." The reasoning formed the basis to formalize the expert knowledge by codifying level descriptions into a set of rules (e.g., Droesen 1996).

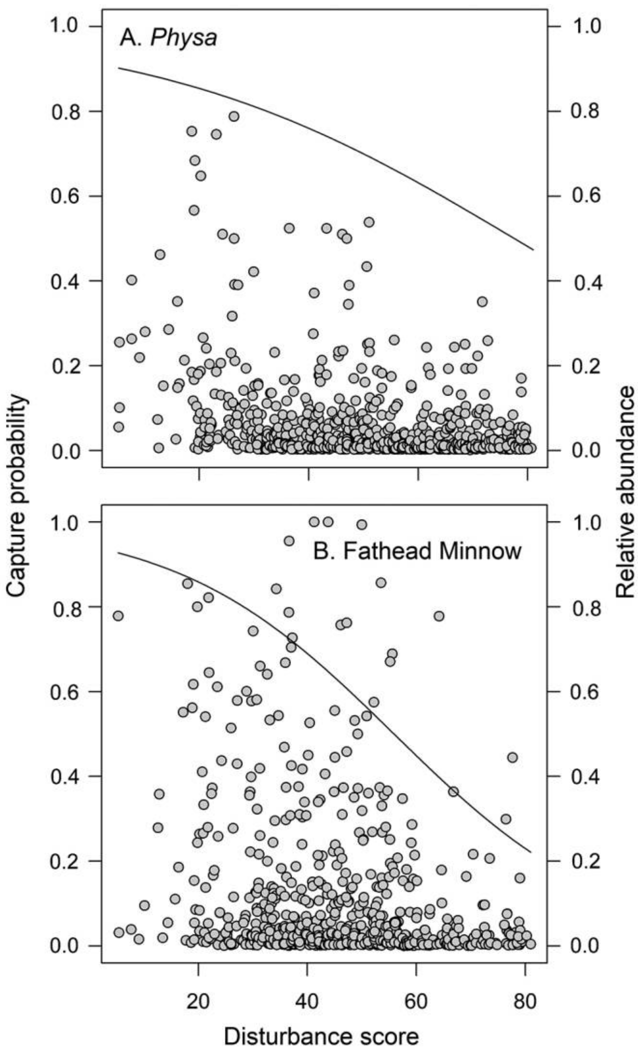

Figure 5. Examples of tolerant taxa, *Physa* (Biological Condition Gradient [BCG] attribute V; tolerant) (A) and Fathead Minnow (BCG attribute Va; highly tolerant) (B). These species occur throughout the disturbance gradient, but with higher probability of occurrence and higher abundances in more stressed sites. See Table 2 for BCG attributes.

Table 4. Examples of macroinvertebrate taxa by Biological Condition Gradient (BCG) attribute group. Assignment to attribute varied between habitats (high-gradient and low-gradient) for some taxa, so number of taxa represents the range of the number of genera assigned to the attribute group among stream types.

BCG attribute	Number of taxa	Example taxa
I Endemic, rare	1	*Goera*
II Highly Sensitive	29–41	*Stempellina*, *Heleniella*, *Ephemerella*, *Paraleuctra*, *Ophiogomphus*, *Parapsyche*, *Diplectron*,, *Lepidostoma*, *Dolophilodes*, *Rhyacophila*
III Intermediate Sensitive	107–148	*Diamesa*, *Tvetenia*, *Hexatoma*, *Plauditus*, *Parapoynx*, *Isoperla*, *Boyeria*, *Amphinemura*, *Pycnopsyche*, *Brachycentrus*, *Limnephilus*
IV Intermediate Tolerant	201–231	Dytiscidae, Ceratopogonidae, *Polypedilum*, *Limonia*, *Perlesta*, *Heptagenia*, *Libellula*, *Hydropsyche*, *Sphaerium*, *Planorbella*
V Tolerant	25–41	Erpobdellidae, *Cricotopus*, *Pseudocloeon*, Corixidae, *Enallagma*, *Caecidotea*, Physidae
VI Nonnative	1	*Corbicula*
x Unassigned	20	Family identifications or unusual taxa; *Chaoborus*, *Zavrelia*, *Didymops*, Nemata

Rule development required discussion and documentation of BCG-level assignment decisions and the reasoning behind the decisions. During this discussion, we recorded: 1) each participant's decision ('vote') for the site; 2) the critical or most important information for the decision, e.g., the number of taxa of a certain attribute, the abundance of an attribute, the presence of indicator taxa; and 3) confounding or conflicting information and how the conflict was resolved for the eventual decision.

After initial site assignment and rule development, we estimated descriptive statistics of the attributes and other biological indicators for each BCG level determined by the panel. These descriptions assisted in review of the rules and their iteration for testing and refinement. The first 2 panel sessions were in-person, 3-d workshops, and subsequent panel sessions were by webinar. The initial panel decisions comprised a preliminary set of decision rules. We quantified the rules in Excel® (versions 2003–2013; Microsoft, Redmond, Washington) workbooks, and calculated BCG level assignments for each sample. We evaluated model performance by comparing model-assigned BCG levels to the panel assignments. Following the initial development phase, the panel tested the draft rules with new data to ensure that new sites were assessed in the same way. Any remaining ambiguities and inconsistencies from the first iterations were resolved.

Table 5. Examples of fish taxa by Biological Condition Gradient (BCG) attribute group. Assignment to attribute varied among stream types for some species, so number of taxa represents the range of the number of species assigned to the attribute group among 7 stream types.

BCG attribute	Number species	Example species
I Endemic, rare	1–9	Blue Sucker, Crystal Darter, Gilt Darter, Greater Redhorse, Lake Sturgeon, Pugnose Shiner, River Redhorse, Shovelnose Sturgeon, Topeka Shiner
II Highly sensitive	6–17	American Brook Lamprey, Blackchin Shiner, Brook Trout, Southern Brook Lamprey, Western Sand Darter
III Intermediate sensitive	15–35	Blacknose Shiner, Burbot, Golden Redhorse, Hornyhead Chub, Shorthead Redhorse, Smallmouth Bass
IV Intermediate tolerant	26–43	Common Shiner, Gizzard Shad, Johnny Darter, Northern Pike, Spotfin Shiner, White Sucker[a]
V Tolerant	5–18	Creek Chub, Brassy Minnow, Brook Stickleback, Central Stoneroller, Sand Shiner
Va Highly tolerant	7–8	Bigmouth Shiner, Bluntnose Minnow, Fathead Minnow, Green Sunfish
VI Sensitive nonnative	3	Brown Trout, Rainbow Trout, Chinook Salmon
VIa Tolerant nonnative	4	Common Carp, Goldfish, Ruffe, Threespine Stickleback
x unassigned	2	Unidentified fish, hybrids

[a] White Sucker is identified tolerant (attribute V) in wadeable streams only.

BCG inference models

The decision models calculated BCG levels directly from the quantified rules by applying fuzzy logic (Zadeh 1965, 2008). Instead of a statistical prediction of expert judgment, this approach directly and transparently converts the expert consensus to automated site assessment. Fuzzy logic is "a precise logic of imprecision and approximate reasoning" (Zadeh 2008). It is directly applicable to environmental assessment and has been used extensively in engineering and environmental applications worldwide (e.g., Castella and Speight 1996, Ibelings et al. 2003, Demicco and Klir 2004, Cheung et al. 2005, Joss et al. 2008).

Fuzzy logic and set theory allows degrees of truth, in contrast to binary truth in classical logic and set theory. For example, one can compare how classical set theory and fuzzy-set theory treat classification of sediment, where sand is defined as particles ≤2.0 mm diameter and gravel is >2.0 mm (Klir 2004). In classical 'crisp' set theory, a particle with diameter = 2.00 mm is classified as sand, and one with diameter = 2.01 mm is classified as gravel. In fuzzy-set theory, both particles have nearly equal membership in both classes (Klir 2004). Measurement error as small as 0.005 mm greatly increases the uncertainty of classification in classical set theory, but in fuzzy-set theory a particle near the boundary would have nearly equal membership in both sets (sand and gravel). Thus, fuzzy sets retain the understanding and knowledge of measurements close to a set boundary, which is lost in classical sets. For further explanation of fuzzy logic, see Klir (2004) or any online tutorial.

To develop the fuzzy inference model, each linguistic variable (e.g., high taxon richness) is defined quantitatively as a fuzzy set (e.g., Klir 2004). A fuzzy set has a membership function in the range of 0 to 1 that determines whether an object is in the set or not in the set. Example membership functions of different sets of taxon richness are shown in Fig. 6A, B. We used piecewise linear functions (i.e., functions consisting of line segments) to assign membership values. If the number of taxa is less than or equal to the lower threshold it has membership of 0, if the number of taxa is greater than or equal to an upper threshold it has membership of 1, and if the number of taxa is between the thresholds, the membership is assigned using a linear interpolation between the lower and upper thresholds. For example, a sample with 30 total taxa would have a membership of ~0.5 in the set 'Moderate number of taxa' and a membership of 0.5 in the set 'High number of taxa' (Fig. 6A).

Assigning membership on the basis of fuzzy-set theory is different from doing so on the basis of classical set theory. Suppose 2 rules determine whether a water body is BCG Level 3: 1) the number of total taxa is high and 2) the number of sensitive taxa is moderate or higher (shaded areas in Fig. 6A, B). If both rules must be true, they are combined with the Boolean AND operator. In fuzzy-set theory, the Boolean AND operator is equivalent to the

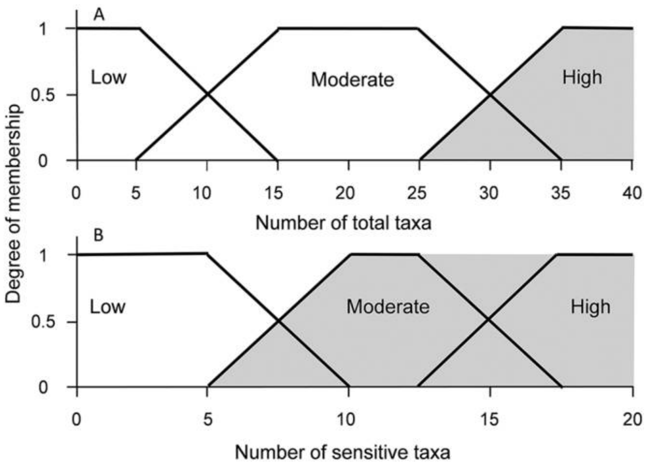

Figure 6. Fuzzy-set membership functions assigning linguistic values to defined ranges for total taxa (A) and sensitive taxa (B). Shaded regions correspond to example rules for Biological Condition Gradient Level 3: "number of total taxa is high" and "number of sensitive taxa is moderate or greater."

minimum membership given by each rule: Level 3 = MIN (total taxa is high, sensitive taxa is moderate or higher). For 32 total taxa and 7 sensitive taxa, fuzzy membership in total taxa is high = 0.6 (Fig. 6A), and fuzzy membership in sensitive taxa is low-moderate to moderate = 0.4 (Fig. 6B). Membership of BCG Level 3 is then 0.4, indicating that the site is "somewhat like Level 3 sites, but not overwhelmingly"; i.e., it is borderline. In the fuzzy-set case, a single additional sensitive taxon raises the membership in BCG Level 3 from 0.4 to 0.6, indicating it is somewhat more like Level 3, but still borderline. In classical set theory, the boundaries of the categories in Fig. 6A, B would be vertical lines. A sample with 7 sensitive taxa would be deemed not in Level 3, but a sample with 8 sensitive taxa would be deemed in Level 3.

If the 2 rules are combined with an OR operator, then either can be true for a site to meet BCG Level 3. In words, we would say, "BCG Level is 3 if total taxa are high OR sensitive taxa are moderate or higher." Classical set theory now yields a value of 'true' if total taxa = 32 and sensitive taxa = 7 (total taxa > 32, therefore, it is true). Fuzzy-set theory yields a membership of 0.6 (maximum of 0.4 and 0.6). In practice, the OR operator is specified only occasionally, when the panel wishes to set up alternative criteria for a certain decision.

In the decision model, rules work as a logical cascade from BCG Level 1 to Level 6. A sample is first tested against the BCG Level 1 rules. If a required rule fails, then the level fails, and the assessment moves down to BCG Level 2, and so on. Required rules are combined with AND operators (i.e., all must be true), and alternate rules are combined with OR operators. Membership in any BCG Level ranges from 0 to 1, and the model requires all membership values to sum to 1. The highest membership is taken as the nominal level, although memberships within 0.2 of each other are considered ties. For example, if the membership of BCG Level 2 is 0.5 and Level 3 is 0.4, then the site is considered to be intermediate between Levels 2 and 3. The output of the model is the nominal BCG level and its membership value and the 2nd (runner-up) BCG level and its membership value.

Because MPCA intended to use the BCG to develop meaningful thresholds for its IBI indexes, the BCG scores were compared to IBI scores from all available biological visits. This analysis consisted of examining box plots and outliers (e.g., sites with high IBI scores, but BCG scores indicating an altered community). The intent of this analysis was not to identify individual visits and bring them in alignment with BCG expectations, but to identify groups of similar communities that were not part of the calibration or test data sets and might require changes to both BCG and IBI models. This effort was parsimonious because too much modification to the models could lead to over-fitting or altering the model from the intent of the panel.

RESULTS

Stress-response relationships and BCG taxa attributes

We examined stress-response scatterplots and estimated maximum likelihood models for taxon occurrence for all taxa with >20 occurrences in the data set (Figs 2, 3A, B, 4A, B, 5A, B, S1, S2). HDS scores were not evenly distributed with relatively few sites with scores <40 (highly altered). An apparent reduction in point density at low-disturbance scores reflects the fact that few sites in the database had such low scores and not necessarily the response of the taxa. The capture probability curve takes the distribution of disturbance scores into account and shows which taxa are tolerant or thrive under disturbed conditions (Figs 2, 3A, B, 4A, B, 5A, B, S1, S2).

Scatterplots that combined abundances of individual taxa on the disturbance gradient with the maximum likelihood models were deemed to be the most useful for identifying attribute groups (Tables 4, 5, Figs 2, 3A, B, 4A, B, 5A, B). Fish species were assigned to attributes separately for each of the 7 fish stream types, and macroinvertebrates were assigned separately to 2 groups: high-gradient and low-gradient streams. Only a few taxa differed in assigned attribute among stream types.

Fish experts identified 2 additional subattributes related to highly tolerant taxa (Table 5). An additional very tolerant classification was created (attribute Va). Separation of the highly tolerant attribute Va fish from the merely tolerant attribute V fish was based on the collective professional experience and judgment of the fish panel. The nonnative fish taxa attribute (VI) was similarly divided into sensitive nonnative salmonids (attribute VI; e.g., Brown Trout and Rainbow Trout) and highly tolerant nonsalmonid, nonnative species (attribute VIa; e.g., Ruffe, Sea Lamprey, Common Carp).

In total, 133 fish taxa and 516 macroinvertebrate taxa were assigned to BCG taxonomic attributes (Tables S1, S2). An additional 53 fish species occurred in MDNR's species list, but were absent from the stream data set and were left unclassified, and 10 fish taxa in the data were left unclassified (family- or genus-level identifications or hybrids considered uninformative). Twenty invertebrate taxa were left unassigned because participants thought information on the taxa was insufficient, or they were relatively unusual in the data set.

Site assignments to BCG levels

The panel was able to reach a majority opinion on the BCG level assignments for all sites reviewed. Some sites required discussion and resolution of disagreement on which of 2 adjacent BCG levels to assign the site. These sites were considered intermediate, with characteristics of both adjacent BCG levels.

The panels were able to distinguish 6 BCG levels (BCG Levels 1–6), but sites that fit Levels 1 (nearly pristine) and 6 (extreme degradation) were rare. The fish panel identified 9 BCG Level 1 sites, but the macroinvertebrate panel identified none. In general, macroinvertebrate experts felt that BCG Level 1 and Level 2 sites were not distinguishable based on macroinvertebrate data only, in part because rare and endemic taxa are poorly identified, their historic distributions are poorly known, and macroinvertebrate sampling methods are inefficient at finding rare and endemic species. Further examination may be necessary to decide whether any sites meet criteria for minimally disturbed (Stoddard et al. 2006). The macroinvertebrate panel identified 9 and the fish panel identified 8 BCG Level 6 samples.

Attributes and BCG levels

We derived metrics (e.g., taxon richness, % taxa, % individuals, dominance) based on BCG attributes and taxonomic groupings (see examples in Figs 7A–F, 8A–F, 9A–D, 10A–E). These box plots were used to help with the selection of metrics for initial model development and for panel review of metrics and rules during subsequent iterations. We developed the BCG using only taxonomic information (attributes I–VI; USEPA 2016) because MPCA's monitoring program does not require collection of information on the other attributes. If available, information from attributes VII–X could be incorporated into the BCG models to improve their performance.

BCG rule development

Panelists followed the descriptions of the BCG levels (Table 1) and gave their reasoning during the deliberations for assigning sites to levels. Rules and reasoning of the panel, whether quantitative or qualitative, were compared to data summaries of the panel decisions (Figs 7A–F,

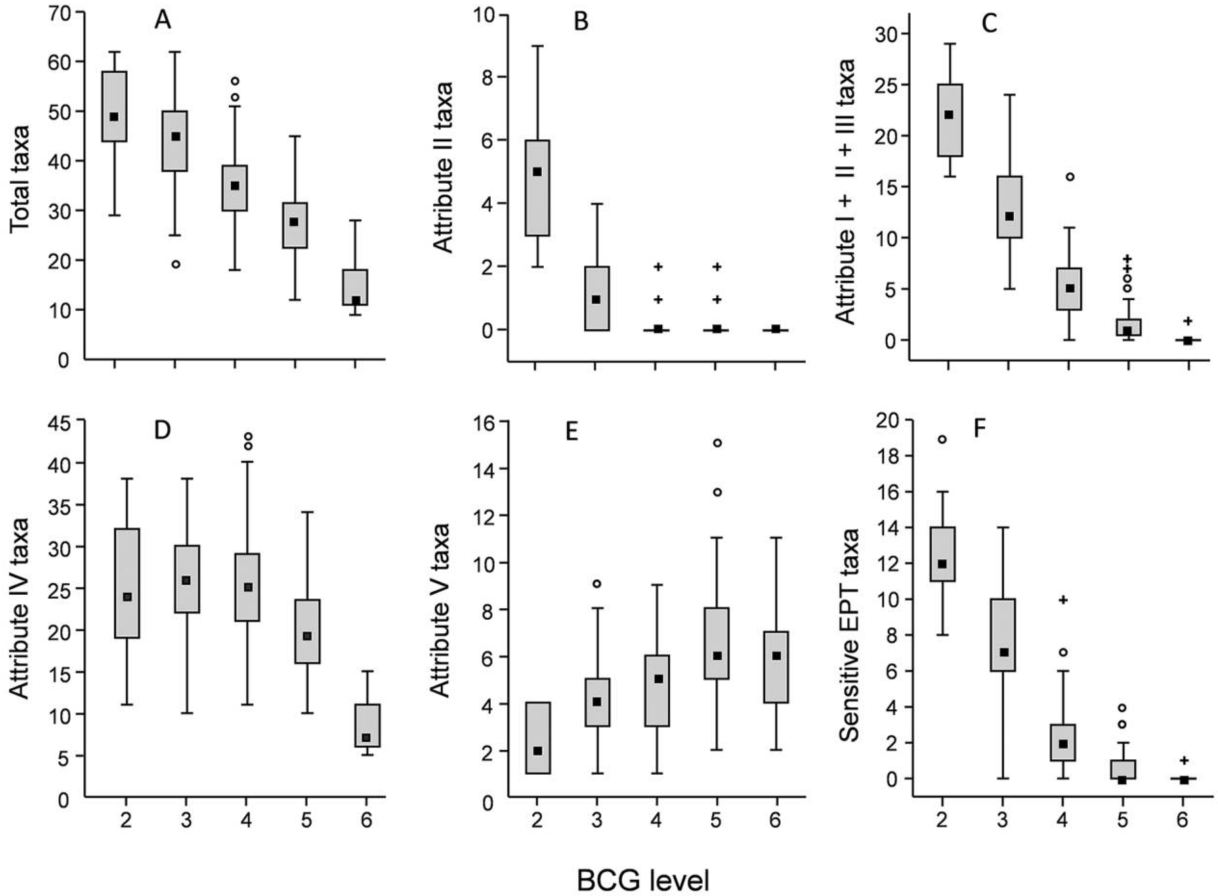

Figure 7. Box-and-whisker plots for the total (A), attribute IV (B), attributes I, II, and III (C), attribute IV (D), attribute V (E), and sensitive Ephemeroptera, Plecoptera, Trichoptera (EPT) (F) number of benthic macroinvertebrate taxa by Biological Condition Gradient (BCG) level. Squares in boxes are medians, boxes are interquartile range (IQR), whiskers are to 1.5 × IQR, circles are outliers up to 3 IQR, and crosses show extreme values > 3 IQR.

8A–F, 9A–D, 10A–E). For example, if the panel identified a moderate number of sensitive taxa for BCG Level 3, then we examined the number of sensitive taxa in samples the panel assigned to BCG Level 3. We then selected a reasonable minimum of the distribution of sensitive taxa in BCG Level 3, say the minimum or a 10th quantile, as the decision threshold. This process was repeated for all rules and attributes identified by the panel as being important to their decisions. Sample sizes for the highest and lowest levels (BCG Levels 1, 2, and 6) were small, and required increased professional judgment from the panel to develop rules.

For a particular attribute or metric, the threshold identified by the panel typically was the 50% membership value in a fuzzy membership function. For example, if the panel identified ">10" sensitive taxa as a requirement for BCG Level 3 (Fig. 7A–F), then 10 taxa would correspond to 50% membership, 5 taxa might correspond to 0% membership, and 15 taxa to 100%. Because number of taxa is always a whole number, this membership function is not continuous. Some rules are non-fuzzy: if a rule requires "≥1" or "presence," then presence receives a membership of 100% and absence receives 0%. Final rules for all 14 assessed stream types are in Tables S3–S8. We include 2 sets of rules here for illustration: riffle–run invertebrate samples (Table 6) and wadeable stream fish samples (Table 7).

Panelists preferred to use taxon richness within the sensitive attributes as the most important criteria for setting site BCG level assignments. Thus, the number of sensitive taxa was most often used to distinguish BCG Level 2 from Level 3 sites. BCG Level 2 should have several highly sensitive taxa (attribute II), but their richness may be reduced or absent in BCG Level 3. All of the BCG Level 1 fish samples had ≥2 attribute I taxa (rare or endemic taxa). Higher BCG levels (1–3) all required some minimum relative abundance or relative richness of sensitive taxa (attributes I–III). In addition, for a site to be considered in Level 1 to Level 3, participants often placed upper limits on the abundance and dominance of tolerant taxa, especially attributes V and Va (for fish). Going further down the gradient, BCG Level 4 typically had a fairly low minimum requirement for sensi-

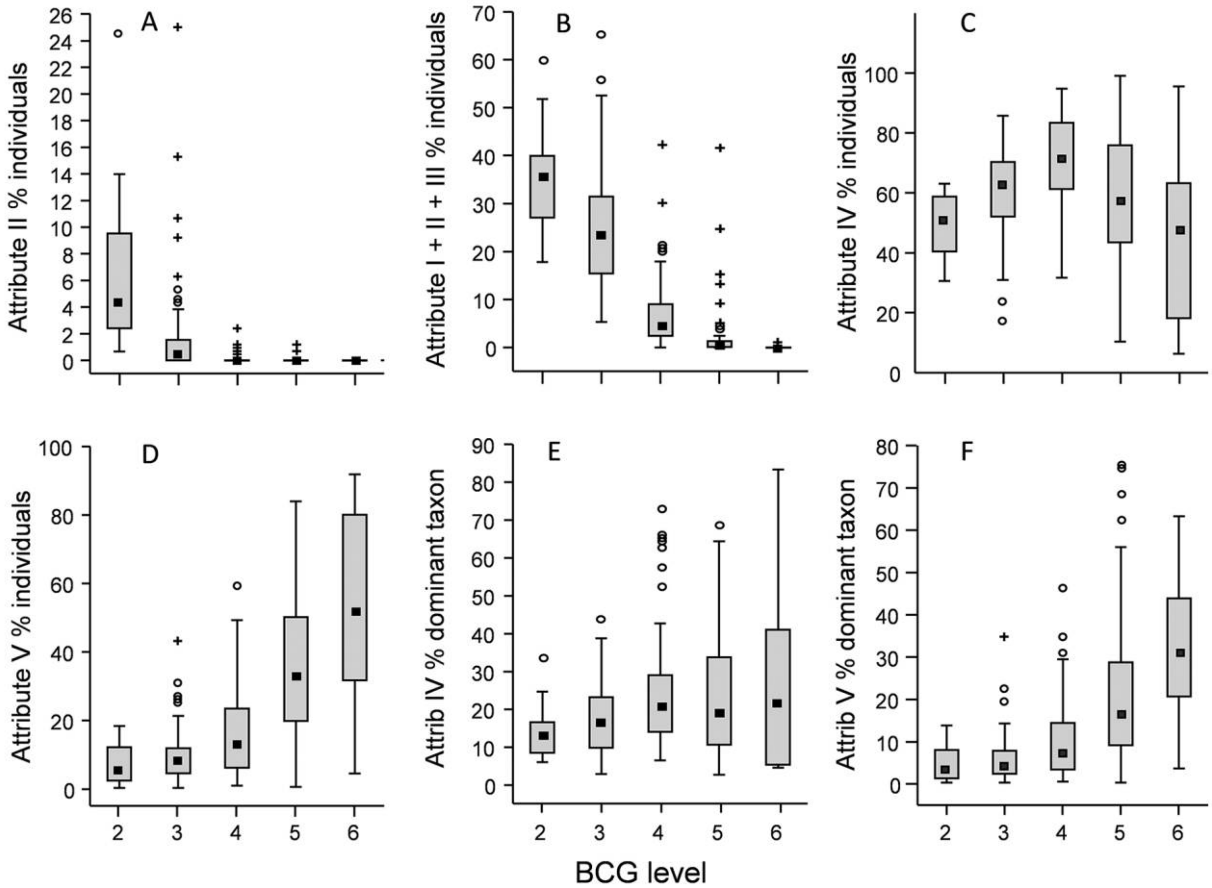

Figure 8. Box-and-whisker plots for the % attribute II (A), attributes I, II, and III (B), attribute IV (C), attribute V (D) individuals and % dominance of attribute IV (E) and attribute V (F) genera of benthic macroinvertebrates by Biological Condition Gradient (BCG) level. See Fig. 7 for explanation of plots.

tive taxa (attribute III), sufficient to show they had not disappeared. BCG Level 5 usually had only requirements of minimum overall richness, and often a maximum dominance (not to be exceeded) of a tolerant taxon. Failure of Level 5 rules result in an assessment of Level 6. The decision patterns described here are consistent with those developed in other states and regions by other panels for invertebrates and fish (see case studies in USEPA 2016).

Rules (Tables 6, 7, S3–S8) were expressed as an inequality, a midpoint, and a range: e.g., ≥20 (15–25). The first number is the midpoint, and the range is in parentheses, where the range describes the linear fuzzy membership function as it increases from 0 to 1 for '≥' and decreases from 1 to 0 for '≤'. Thus, for a rule expressed as ≥20% (15–25), the given membership is 0 at a metric value ≤15%; rises linearly to 1 at a metric value of 25%; and remains 1 for values >25%. The membership is 0.5 at the midpoint of 20%.

Some rule sets included alternatives; i.e., 2 or 3 alternative rules may exist for a certain BCG level (e.g., BCG Level 3 in Table 6, Levels 4 and 5 in Table 7). At least one of the alternatives must be true for the site to be assigned to that level. Alternatives usually reflected a trade-off specified by the panel. For example, a high number of total taxa could offset a low proportion of sensitive taxa, and vice versa. Rules *within* each alternative are joined by AND operators, and the 2 or 3 alternatives are then joined by OR operators to assign level.

Model performance

To evaluate the performance of the quantitative decision model, we assessed the number of samples where the BCG decision model's nominal level exactly matched the panel's median (exact match) and the number of samples where the model predicted a BCG level that differed from the median expert opinion (mismatch samples). For the mismatched samples, we examined the size of the difference between the BCG level assignments.

The model output is in terms of relative membership (0–100%) of a site among BCG levels, where memberships of all levels must sum to 100%. Model output could yield ties between adjacent levels, or a majority could be assigned to 1 level over ≥1 other levels. As with the quanti-

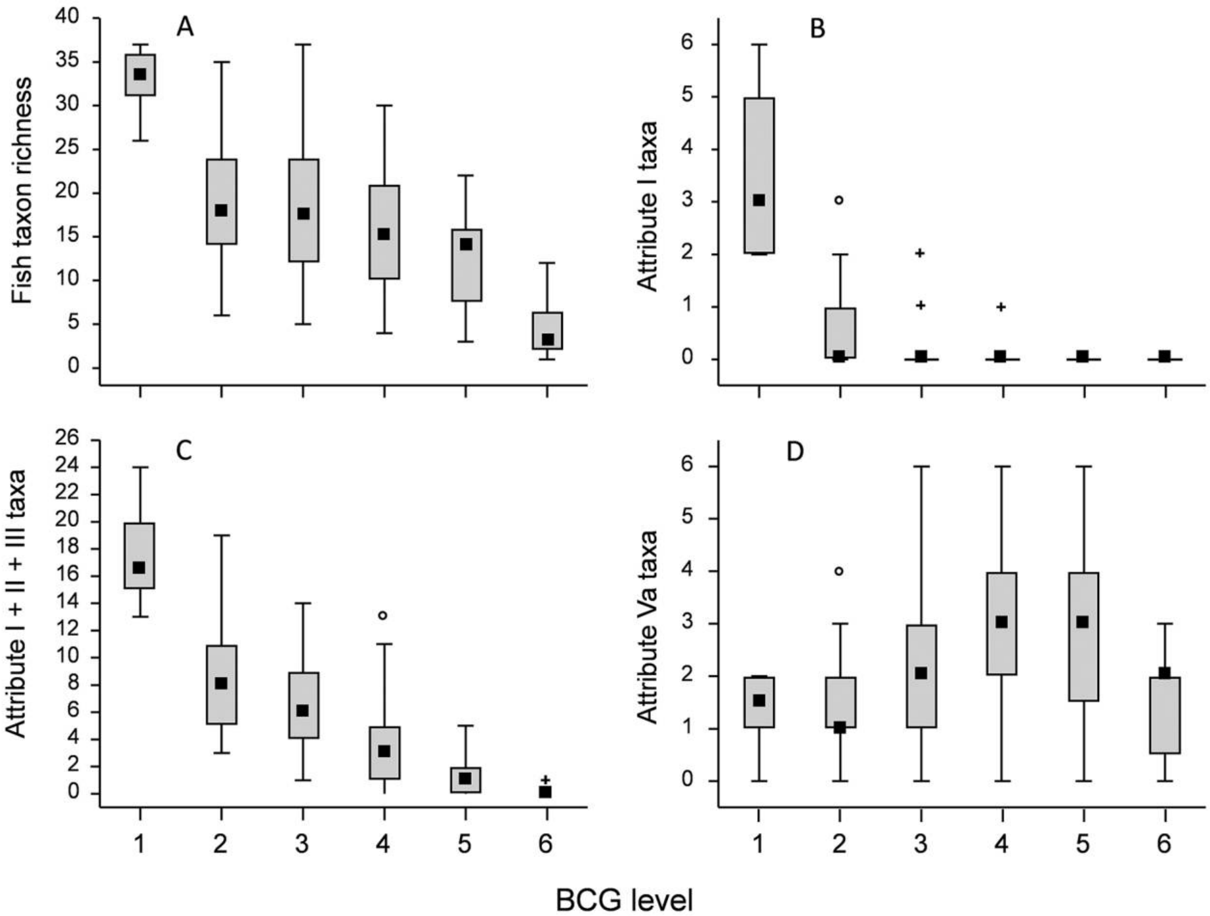

Figure 9. Box-and-whisker plots for the total (A), attribute I (B), attributes I, II, and III (C), and attribute Va (D) number of fish taxa by Biological Condition Gradient (BCG) level. See Fig. 7 for explanation of plots.

tative model, panelists' site ratings could be split among BCG levels.

To estimate concurrence between the quantitative model and the panel, we assigned scores as clear majority or ties and near-ties based on the panelists' votes and the model membership outcomes. We assigned ties and near-ties where either the model or the panel was divided. For model ties, nearly equal membership was present in 2 BCG levels (e.g., membership of 0.4–0.6 in BCG Level 2 and membership of 0.6–0.4 in BCG Level 3). Panelist ties were site ratings where a single vote could have flipped the decision (e.g., 4–4 or 5–4 decisions).

If either the BCG model assigned a tie that did not match with the panelist consensus, or vice-versa, we assigned a difference of ½ BCG level. For example, if the model assignment was a BCG Level 2–3 tie and panelist consensus was BCG Level 2, the model was considered to be off by ½ BCG level; more specifically, the model rating was a ½ BCG level worse than the panelists' consensus. To avoid cutting the differences too finely, we considered mismatches by units of only ½ BCG level. These units were: match (i.e., both panel and model a clear majority for the same level or the same tie); ≤½ level (i.e., panel and model mismatch by ≤½ BCG level); ≤1 level (i.e., panel and model mismatch ½ but ≤1 BCG level); and so on.

Model performance is summarized in Tables 8 and 9, which show the number and % model assessments compared to panel assessments. The panel did not consider a ½-level mismatch with their consensus to be a meaningfully different assessment, and a ½ level was similar to the spread in ratings among panel members. Thus, the panel was unwilling to adjust ratings or to modify rules for small mismatches. On average, the macroinvertebrate models were 89% accurate in replicating the panel assessments within ½ BCG level, and the fish models were 86% accurate. The fish model had 2 mismatches >1 BCG level.

We compared BCG model performance on all sites to IBI models, which had been developed independently. Neither IBI model nor BCG model was regarded as objective truth. Rather, the comparison was used to identify situations where, in the expert opinion of the panel, either or both models might need modifications. Overall, the IBI and BCG models corresponded to each other, but interquartile ranges did overlap between adjacent BCG levels

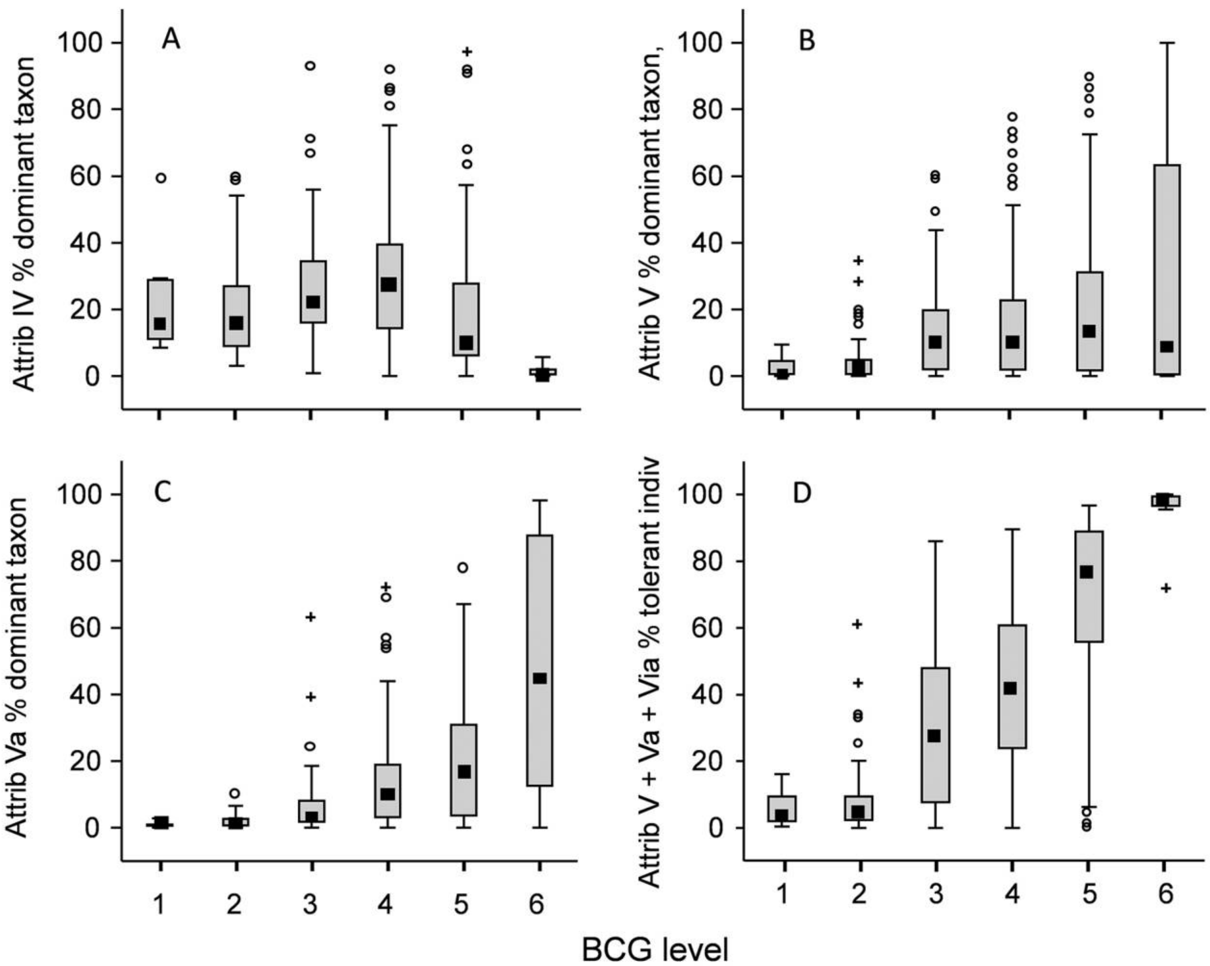

Figure 10. Box-and-whisker plots for the % dominance of attribute II (A), attribute V (B), and attribute Va (C) fish, and % tolerant (attributes V, Va, and VIa) individuals (D) by Biological Condition Gradient (BCG) level. See Fig. 7 for explanation of plots.

(Figs 11A–G, 12A–G). In some stream types, the distribution of IBI scores for BCG Level 6 appeared anomalous. Differences between the 2 models often were the result of differences in the scoring approaches. For example, with the IBIs, a biological sample might score extremely poorly for a single metric, but because the final score is a sum of multiple metric scores, the final score could still be high or intermediate if other metrics score high, a phenomenon known as "eclipsing" (Suter 1993).

The exercise also identified situations where the panel thought the BCG rules were too stringent, and the rules subsequently were relaxed. These changes included modifying the thresholds for some metric criteria or, in some cases, addition of alternate criteria (e.g., BCG Level 3 in Table 6). The alternate criteria provide multiple paths to a higher BCG level score for a sample and account for the diversity of healthy communities within a stream type. The rule changes improved the applicability of the BCG models beyond the population of the sites used in the model development and testing efforts. This exercise also indicated where changes should be made to the IBI models. The process identified a small number of samples with poorly scoring biological communities in relatively undisturbed watersheds. These streams were often wetland-influenced streams, and new IBI and BCG models are needed to measure biological condition appropriately for this type of stream.

Fish-invertebrate assemblage comparison

An issue of interest to managers is whether fish and macroinvertebrate assemblages yield the same results, and whether both must be monitored. We examined BCG assessments by the panel for a set of sites with both fish and benthic macroinvertebrate samples that were sampled in the same calendar year (typically within 1–3 mo). The maximum difference found was 3 BCG levels in 2 rivers where the fish were rated Level 2 but the invertebrates were rated Level 5. Both assemblages were rated at ≤1 BCG

Table 6. Decision rules for macroinvertebrate assemblages in high-gradient streams (riffle–run habitat). Rules show the midpoint and ranges (in parentheses) of fuzzy membership functions (see Fig. 6). N is the number of sites at the indicated Biological Condition Gradient (BCG) level and stream type in the calibration data set. 'Alt' designation in rules identifies alternative rule sets for a particular stream type and BCG Level (see text for details). EPT = Ephemeroptera, Plecoptera, Trichoptera; '= Alt 1' indicates the rule is the same as given under Alt 1 for this metric; n/a = not applicable.

	Northern forest streams, high-gradient		Southern streams, high-gradient	
Metric	Alt 1	Alt 2	Alt 1	Alt 2
BCG Level 2	$N = 2$		$N = 0$[a]	
Total taxa	≥40 (35–45)	n/a	≥40 (35–45)	n/a
Attribute I+II taxa	>3 (2–5)	n/a	>3 (2–5)	n/a
Attribute I+II+III % taxa	≥50% (45–55)	n/a	≥50% (45–55)	n/a
Attribute I+II+III % individuals	≥30% (25–35)	n/a	≥30% (25–35)	n/a
Attribute V % individuals	≤10% (7–13)	n/a	≤10% (7–13)	n/a
Sensitive EPT taxa	>11 (9–14)	n/a	>11 (9–14)	n/a
BCG Level 3	$N = 17$		$N = 8$	
Total taxa	≥30 (25–35)	≥45 (40–50)	≥30 (25–35)	≥45 (40–50)
Attribute I+II+III % taxa	≥20% (15–25)	≥15% (10–20)	≥20% (15–25)	≥10% (7–13)
Attribute I+II+III % individuals	≥10% (7–13)	≥5% (3–7)	≥15% (10–20)	≥5% (3–7)
Attribute IV dominance	≤25% (20–30)	= Alt 1	n/a	n/a
Attribute V % individuals	n/a	n/a	≤20% (15–25)	= Alt 1
Attribute V dominance	≤35% (30–40)	= Alt 1	≤10% (7–13)	= Alt 1
Sensitive EPT taxa	>3 (2–5)	= Alt 1	>3 (2–5)	= Alt 1
BCG Level 4	$N = 9$		$N = 19$	
Total taxa	≥20 (16–24)	n/a	≥20 (16–24)	≥30 (25–35)
Attribute I+II+III % taxa	≥10% (7–13)	n/a	≥5% (3–7)	Present
Attribute I+II+III % individuals	Present	n/a	≥5% (3–7)	Present
Attribute V % individuals	≤25% (30–40)	n/a	≤25% (30–40)	≤40% (35–45)
Attribute V dominance	≤25% (20–30)	n/a	≤20% (15–25)13	= Alt 1
Sensitive EPT	Present	n/a	Present	= Alt 1
BCG Level 5	$N = 2$		$N = 20$	
Total taxa	>13 (11–16)	≥20 (16–24)	>13 (11–16)	≥20 (16–24)
Attribute II+III+IV % taxa	n/a	n/a	n/a	≥50% (45–55)
Attribute V % taxa	≤40% (35–45)	≤50% (45–55)	≤40% (35–45)	n/a
Attribute V dominance	≤60% (55–65)	= Alt 1	≤60% (55–65)	n/a
BCG Level 6	$N = 0$		$N = 0$	

[a] BCG rules for southern streams, high-gradient Level 2 provisionally set to same criteria as northern forest streams, high-gradient

level apart in 83% of the sample sets (Table 10). The macroinvertebrate panel was slightly more stringent than the fish panel: no invertebrate samples were rated BCG Level 1, and slightly fewer Levels 2 and 3 ratings were given by the macroinvertebrate panel than by the fish panel. More large differences of ≥2 levels (Table 10) occurred at river than at wadeable stream sites. Fish and invertebrates were rated at ≥2 BCG levels apart in 40% of large river sites (non-wadeable; drainage area > 1300 km^2) but in only 9% of wadeable sites. The 2 assemblages respond to different stressors, so we would not expect a perfect correlation between ratings based on macroinvertebrates and on fish. Both assemblages are sampled and assessed because of their different responses.

DISCUSSION

Recent developments of environmental assessment using professional judgment have shown that experts are highly concordant in their ratings of marine benthic macroinvertebrates (Weisberg et al. 2008, Teixeira et al. 2010), marine sediment quality (Bay et al. 2007, Bay and Weisberg 2010), and fecal contamination (Cao et al. 2013). In the pilot BCG studies (USEPA 2016), aquatic biologists have

Table 7. Decision rules for fish assemblages in wadeable streams. Rules show the midpoint and ranges (in parentheses) of fuzzy membership functions (see Fig. 6). N is the number of sites at the indicated Biological Condition Gradient (BCG) level and stream type in the calibration data set. 'Alt' designation in rules identifies alternative rule sets for a particular stream type and BCG level (see text for details). '= Alt 1' indicates the rule is the same as given under Alt 1 for this metric; n/a = not applicable.

	Southern streams			Northern streams	
Metric	Alt 1	Alt 2	Alt 3	Alt 1	Alt 2
BCG Level 1		$N = 0$[a]		$N = 0$[a]	
Total taxa	≥30 (25–35)	n/a	n/a	≥30 (25–35)	n/a
Attribute I endemic taxa	Present	n/a	n/a	Present	n/a
Attribute I+II taxa	>3 (2–5)	n/a	n/a	>3 (2–5)	n/a
Attribute I+II+III % taxa	≥50% (45–55)	n/a	n/a	≥50% (45–55)	n/a
Attribute I+II+III % individuals	≥30% (25–35)	n/a	n/a	≥30% (25–35)	n/a
Tolerant % individuals (V+Va+VIa)	≤5% (3–7%)	n/a	n/a	≤5% (3–7%)	n/a
BCG Level 2		$N = 1$		$N = 8$	
Total taxa	≥20 (16–24)	n/a	n/a	>13 (11–16)	n/a
Attribute I+II+III total taxa	≥8 (6–10)	n/a	n/a	n/a	n/a
Attribute I+II+III % taxa	≥40% (35–45)	n/a	n/a	≥30% (25–35)	n/a
Attribute I+II+III % individuals	≥10% (7–13)	n/a	n/a	≥10% (7–13)	n/a
Attribute Va or VIa dominance	n/a	n/a	n/a	≤10% (7–13)	n/a
Tolerant % individuals (V+Va+VIa)	n/a	n/a	n/a	≤35% (30–40)	n/a
Highly tolerant % individuals (Va+VIa)	≤20% (15–25)	n/a	n/a	n/a	n/a
BCG Level 3		$N = 4$		$N = 10$	
Total taxa	>13 (11–16)	n/a	n/a	>13 (11–16)	n/a
Attribute I+II+III % taxa	≥10% (7–13)	n/a	n/a	≥25% (20–30)	n/a
Attribute I+II+III % individuals	≥5% (3–7)	n/a	n/a	≥5% (3–7)	n/a
Attribute Va or VIa dominance	≤20% (15–25)	n/a	n/a	≤10% (7–13)	n/a
Highly tolerant % individuals (Va+VIa)	≤40% (35–45)	n/a	n/a	≤20% (15–25)	n/a
BCG Level 4		$N = 10$		$N = 15$	
Total taxa	≥8 (6–10)	≥20 (16–24)	n/a	≥8 (6–10)	= Alt 1
Attribute I+II+III % taxa	Present	n/a	n/a	≥5% (3–7)	n/a
Attribute 1+ 2+3 % individuals	≥0.5% (0–1)	n/a	n/a	Present	n/a
I+II+III+IV % individuals	n/a	n/a	n/a	n/a	≥70% (65–75)
Attribute I+II+III+IV % taxa	n/a	n/a	n/a	n/a	≥50% (45–55)
Attribute Va or VIa dominance	≤50% (45–55)	= Alt 1	n/a	≤30% (25–35)	≤20% (15–25)
Tolerant % individuals (V+Va+VIa)	≤70% (65–75)	= Alt 1	n/a	n/a	n/a
Highly tolerant % individuals (Va+VIa)	≤60% (55–65)	= Alt 1	n/a	≤60% (55–65)	n/a
BCG Level 5		$N = 18$		$N = 4$	
Total taxa	≥5 (3–7)	>13 (11–16)	≥20 (16–24)	≥3 (1–5)	n/a
Attribute I+II+III % taxa	n/a	Present	n/a	n/a	n/a
Attribute I+II+III+IV % taxa	≥10% (7–13)	n/a	≥20% (15–25)	≥15% (10–20)	n/a
Attribute Va or VIa dominance	≤50% (45–55)	n/a	n/a	<65–75%	n/a
Highly tolerant % individuals (Va+VIa)	≤70% (65–75)	n/a	n/a	n/a	n/a
BCG Level 6		$N = 2$		$N = 0$	

[a] BCG rules for Level 1 provisionally set to same criteria as Prairie Rivers (Table S4).

come to very tight consensus on the descriptions of individual levels of the BCG and on the BCG level assigned to individual sites. The Minnesota BCG reported here confirms the concordance among experts.

The conceptual model of the BCG was derived from experience of working aquatic ecologists from across the US (Davies and Jackson 2006). Development of the quantitative BCG requires quantitative mapping of biological infor-

Table 8. Performance of Biological Condition Gradient (BCG) quantitative macroinvertebrate models. 'Better' and 'worse' indicate model assessment of stream condition compared to panel (e.g., 'better' if model assessed BCG Level 2, but panel assessed BCG Level 3). N = number of comparisons in category, % = % of comparisons in category.

		Quantitative model performance					
Invertebrate stream type	Type	1 better	0.5 better	Match	0.5 worse	1 worse	Total
Northern forest rivers	N	2	2	26	2	5	37
	%	5%	5%	70%	5%	14%	
Prairie and southern rivers	N	0	4	21	3	1	29
	%	0%	14%	72%	10%	3%	
Northern forest high-gradient	N	1	1	27	3	5	37
	%	3%	3%	73%	8%	14%	
Northern forest low-gradient	N	2	1	28	2	2	35
	%	6%	3%	80%	6%	6%	
Southern high-gradient	N	1	2	35	5	2	45
	%	2%	4%	78%	11%	4%	
Southern forest low-gradient	N	2	1	29	3	1	36
	%	6%	3%	81%	8%	3%	
Prairie low-gradient	N	3	2	44	0	3	52
	%	6%	4%	85%	0	6%	
Total	N	11	13	210	18	19	271
	%	4%	5%	77%	7%	7%	

mation into the conceptual and theoretical model. The BCG is calibrated using a data set, but also requires ecological considerations with wide expert agreement from biologists familiar with the resources. The result is intended to be more general than a regression analysis of biological response to stressors. The BCG uses universal attributes (in this application, only the taxonomic attributes I–VI) that are intended to apply in all regions. Specifics of the attributes (taxon membership, attribute groups indicating good, fair, poor, etc.) do vary across regions and stream types, but the attributes themselves and their importance are consistent. The BCG requires descriptions of the levels from pristine to degraded. Documentation of the rationale for making BCG level determinations (i.e., the rules) provides the foundation for building a robust quantitative model and ensures that future information and discoveries can be related back to the level descriptions.

The approach requires substantial time and effort from the expert panel, but does it also require a rich database? We think the BCG calibration itself can be done with a smaller data set. Stress–response analysis benefits from a large database because we generally require a minimum of 20 occurrences of a taxon to develop the stress–response model. Other sources of tolerances for attribute assignments in the absence of stress–response analysis include existing literature and panelists' experience with the taxa. Early BCG calibrations were successful with 50 to 100 sites assessed by the panel, and stress–response was not used in those efforts (e.g., case studies in USEPA 2016). As a general rule, ≥30 sites in each stream type and perhaps as few as 20 is sufficient for rule development.

In a critique of ecosystem health and indexes, Suter (1993) pointed out technical weakness of common indexes. Weaknesses include: 1) *ambiguity*: one cannot tell why an index value is high or low (although individual metric values will reveal it); 2) *eclipsing*: a high metric value balances a low metric value, with a resulting inappropriate score (site is better or worse than its score indicates); and 3) *arbitrary combining functions*: most multimetric indexes (and observed/expected taxon ratios) are the sum of the component metrics (or component reference taxa), with no weighting or other combining function, nor consideration of why or why not to do so (Suter 1993). Eclipsing is one consequence of arbitrary equal weighting and summing. In the BCG rule-based method, weighting and combining functions are stated and not arbitrary. For example, a rule for a BCG level may require a certain number of sensitive taxa. If a site has too few sensitive taxa, it will be rated at a lower level because the sensitive taxa rule failed. Rules prevent ambiguity (i.e., we know why it failed), eclipsing (i.e., a high value in another attribute or metric does not change the decision, unless a rule specifically allows it), and the combining function for the rules is not arbitrary (i.e., transparent and established by the panel).

We do not suggest that the BCG is a panacea for all current issues in bioassessment. It has distinct disadvantages in development and acceptance. For example, the BCG is labor-intensive to develop, requires a panel of experts who are knowledgeable about local water bodies and biota. It can-

Table 9. Performance of Biological Condition Gradient (BCG) quantitative fish models. 'Better' and 'worse' indicate model assessment of stream condition compared to panel (e.g., 'better' if model assessed BCG Level 2, but panel assessed BCG Level 3). N = number of comparisons in category, % = % of comparisons in category.

		Quantitative model performance								
Fish stream type	Type	1.5 better	1 better	0.5 better	Match	0.5 worse	1 worse	1.5 worse	2 worse	Total
Northern rivers	N	0	4	2	36	4	1	0	0	47
	%	0%	9%	4%	77%	9%	2%	0	0%	
Southern rivers	N	0	5	4	52	10	4	0	0	75
	%	0%	7%	5%	69%	13%	5%	0	0%	
Northern streams	N	1	1	3	22	8	1	0	1	37
	%	3%	3%	8%	59%	22%	3%	0	3%	
Northern headwaters	N	0	2	4	19	2	3	0	0	30
	%	0%	7%	13%	63%	7%	10%	0	0%	
Southern streams	N	0	4	1	23	2	5	0	0	35
	%	0%	11%	3%	66%	6%	14%	0	0%	
Southern headwaters	N	0		1	25	3	3	0	0	32
	%	0%	0%	3%	78%	9%	9%	0	0%	
Low-gradient streams	N	0	2	1	26	1	2	0	0	32
	%	0%	6%	3%	81%	3%	6%	0	0%	
Total	N	1	18	16	203	30	19	0	1	288
	%	0.3%	6%	6%	70%	10%	7%	0	0%	

not be developed and calibrated by an individual with a data set and a computer. Broad acceptance of the BCG may be problematic. Many scientists and managers sometimes implicitly assume that continuous, quantitative models are somehow better than expert consensus. We contend that this assumption is untested, and may be an unfounded personal bias.

Decision analysis

To develop the fuzzy decision analysis system, we needed a set of rules to which we could apply fuzzy logic. The greatest single strength of the fuzzy-model approach may be development of a set of transparent rules that can, in principle, be followed by anyone making a decision on a site. Fuzzy-model rules may seem exotic to those not familiar with the approach, but they are fully laid out and are not hidden in a statistical model or in artificial machine learning. Experts can describe the classes of the BCG in a very general way, but without the specific rules and their combination, their decisions cannot be replicated and the rules cannot be modified effectively as new knowledge is gained.

The fuzzy-rule model replicates expert judgment by direct application of rules. It is only as good as the rules themselves. Experts also make errors, so an iterative process is required for rule development to correct inconsistencies, elicit hidden rules, or recalibrate incorrect membership functions. The fuzzy model does not require a statistical model to predict the expert panel decisions. If one accepts the expert consensus and rules of the BCG, then a fuzzy-model approach is the best way we know to automate it.

The rules have no requirement for linearity or monotonicity of metrics or attributes. For example, a linguistic rule that captures subsidy–stress (e.g., Odum et al. 1979) is permissible, such as "If taxon richness is high and abundance is high, then BCG level is $\leq$3." Moderate taxon richness may indicate very good conditions and fair or poor conditions and could be problematic in monotonic applications of taxon richness to condition. Most biotic indexes (e.g., IBI and RIVPACS models; e.g., Barbour et al. 1999) require monotonic responses of component metrics.

Like the BCG, a fuzzy-decision analysis approach has a disadvantage in acceptance. For example, an unfounded linguistic bias exists among American English-speakers against the term "fuzzy" in any scientific context. This bias has resulted in slower acceptance of fuzzy logic systems in English-speaking countries, especially in the USA, than elsewhere because the word 'fuzzy' has colloquial meaning in the USA (fuzzy thinking, warm and fuzzy). Prominent English-speaking scientists revealed their linguistic bias when criticizing fuzzy theory (see quotes in Zadeh 2008). In continental Europe and Japan, fuzzy logic systems are widely used in engineering and decision analysis, including ecological applications (e.g., Ibelings et al. 2003), because

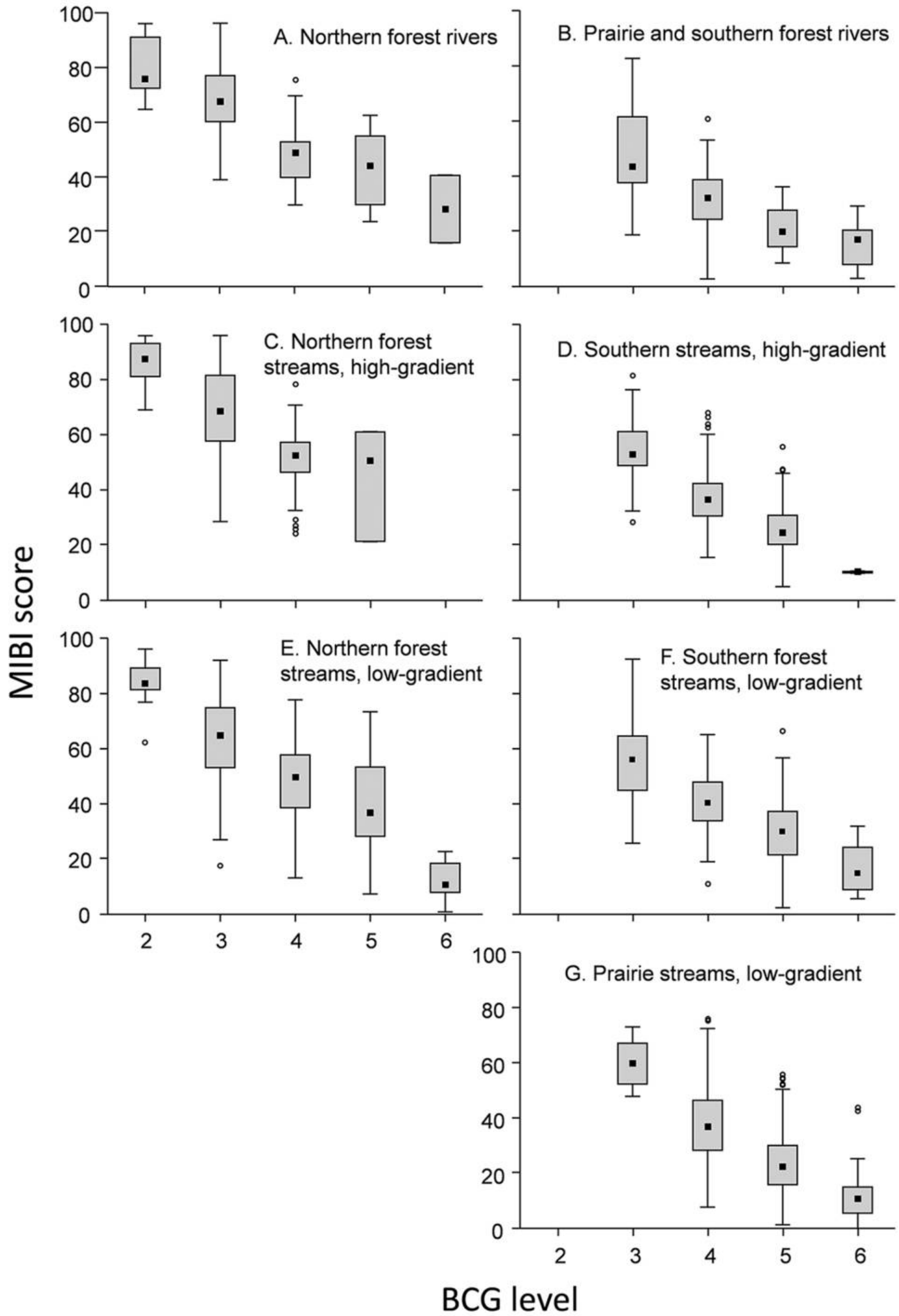

Figure 11. Frequency distributions of Macroinvertebrate Index of Biological Integrity (MIBI) scores by Biological Condition Gradient (BCG) level for northern forest rivers (A), prairie rivers (B), high- (C) and low-gradient (E) gradient northern forest streams, high- (D) and low-gradient (F) southern forest streams, and low-gradient prairie streams (G) in Minnesota at sites sampled from 1996–2011. See Fig. 7 for explanation of plots.

of greater economy of development with respect to nonlinear responses, and because the English word fuzzy has no colloquial connotations in other languages.

We measure things on continuous scales (e.g., pH) or as whole numbers (e.g., counts of taxa), but most interpretations and decisions are binary or categorical. Management and public communication require assessments such as 'no impact', 'slight impact', or 'severe impact'; or decisions such as 'no action' or 'reduce phosphorus by 50%'. Statements such as '5.8 mg/L O_2' or '29 insect species' are neither decisions nor interpretations. Fuzzy-decision systems are an explicit and transparent bridge between continuous

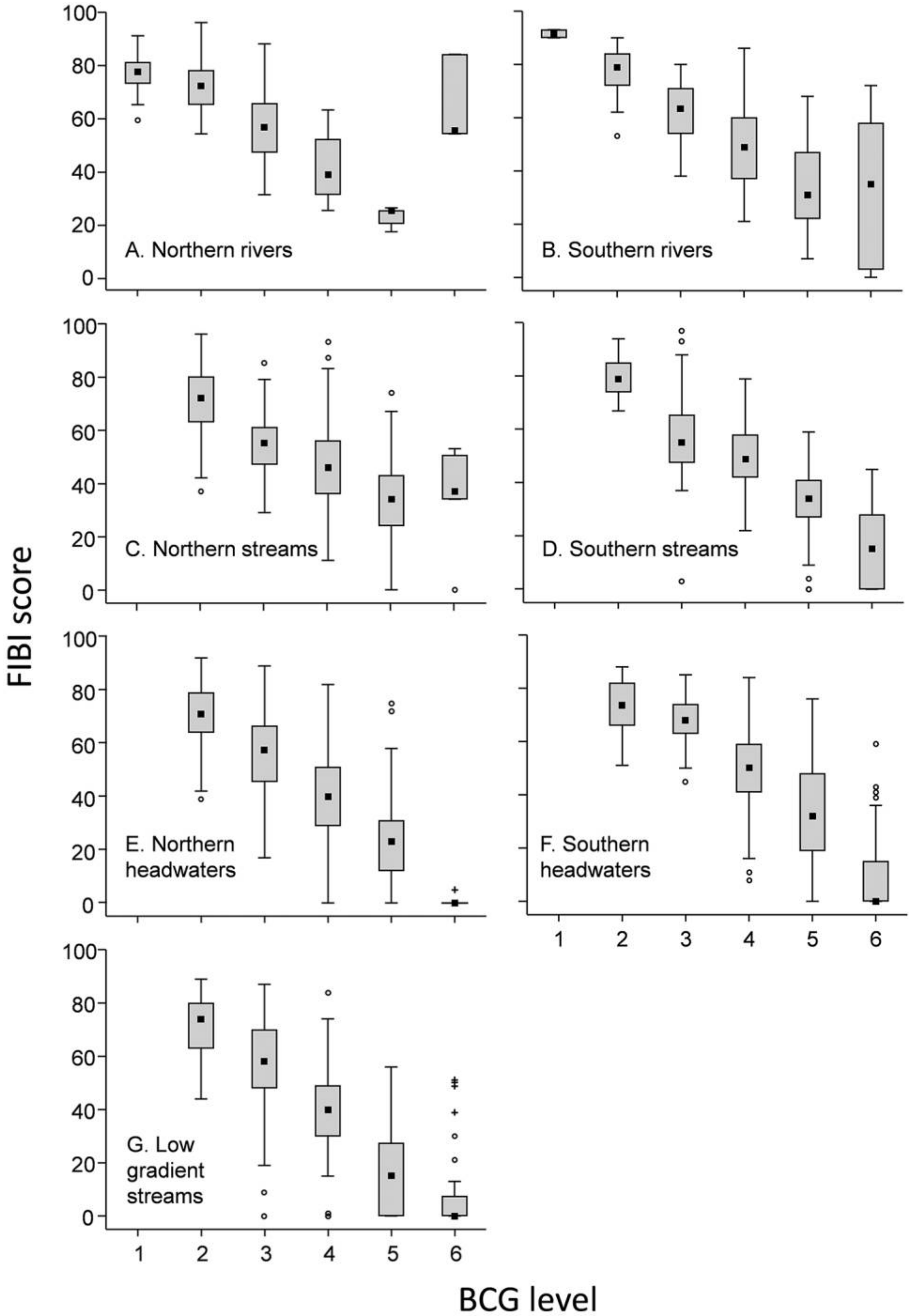

Figure 12. Frequency distributions of Fish Index of Biological Integrity (FIBI) scores by Biological Condition Gradient (BCG) level for northern rivers (A), southern rivers (B), northern streams (C), southern streams (D), northern headwaters (E), southern headwaters (F), and low-gradient streams (G) in Minnesota at sites sampled from 1996–2011. See Fig. 7 for explanation of plots.

measurements and interpretation and management decisions that are categorical (Silvert 2000).

BCG and multimetric IBIs

The BCG and IBI (Figs 10, 11) results were similar, which was not unexpected because they use the same field data sets. Moreover, the fundamental concept of both is that aquatic systems deviate from natural conditions (embodying biological integrity) with increasing anthropogenic stress. However, the development process differed for the models and, therefore, they do not produce identical results. The BCG concept and methods address some issues that multimetric IBI models cannot. The BCG categorizes biological communities in terms of naturalness, whereas the full range of multimetric IBI scores may reflect only avail-

Table 10. Differences in assessment of Biological Condition Gradient (BCG) level for fish and invertebrate communities at sites where both assemblages were sampled in the same calendar year and assessed by the panels ($N = 76$).

Variable	Fish more natural			Same	Invertebrates more natural		
BCG difference	3	2	1	0	1	2	3
N	2	9	22	25	16	2	0

able conditions. The BCG weights metrics and rules according to the panel's judgments, whereas multimetric IBI indexes weight all metrics equally in the total score. The BCG allows for nonlinear or modal responses in the attributes whereas multimetric IBI metrics are monotonic.

Management: aquatic life uses

The Minnesota BCG models are promising as a basis for developing decision criteria or biological criteria for Aquatic Life Uses (ALUs). In the USA, the terms 'Use', 'Designated Uses', and 'Aquatic Life Use' have specific meanings for water-quality management in the context of the CWA. A state defines the uses for its waters and develops physical, chemical, and biological criteria to protect those uses. Designated Uses are the water-quality goals for a specific water body and identify the functions and activities that are supported by a state-defined level of water quality. Water-quality standards are reviewed periodically based on new information that may indicate change in appropriateness of use and changes in what might have been considered irreversible.

Designated Uses also include potential quality or condition that may not be attained currently, but could be attained with appropriate controls or restoration. Thus, ALUs can be set according to the biological potential of water bodies, rather than their current condition. For example, infrastructure is not always irreversible, but it can be modified to reduce stresses on water bodies. The BCG may be more robust than current indexes because it allows for nonlinear responses, and has requirements for combinations of metric values in the condition levels.

The BCG models have been used to refine Minnesota's designated uses known as Tiered Aquatic Life Uses (TALUs; Bouchard et al. 2016, MPCA 2014a). TALUs are refined ALUs that articulate the goal for a water body better than a single one-size-fits-all ALU (e.g., Yoder and Rankin 1995, Bouchard et al. 2016). In Minnesota, the BCG was used to develop biological criteria for TALUs and to address differences in the current condition of streams across the state (Bouchard et al. 2016). For example, the prairie regions in Minnesota have been highly altered, resulting in few if any sites that meet the requirements for minimally disturbed reference sites. This situation poses challenges when the typical reference condition approach is used because minimally disturbed streams are needed to establish benchmarks (i.e., biological criteria) for ALUs. The BCG was used as a universal yardstick to set consistent and protective biological criteria across a diverse landscape (Bouchard et al. 2016). It also aligned biological criteria with the narrative language established by the CWA with the proposed TALU narratives. Levels of the BCG are not a priori equivalent to TALUs or water-quality criteria, although a given criterion could be set to a level of the BCG as a policy decision. The BCG is a measurement yardstick, and it does not express policy decisions and breakpoints for designated uses.

The BCG provides a powerful approach for an operational monitoring and assessment program, for communicating resource condition to the public, and for management decisions to protect or remediate water resources. If formalized properly, any person with data can follow the rules to obtain the same level assignments as the group of experts. This property makes the actual decision criteria transparent to stakeholders. Description of BCG Levels 1 and 2 in the BCG process establishes a fixed natural reference (which may no longer exist) to prevent shifting baselines. Understanding of the natural baseline may be modified with new and better information on historic conditions, but both original and modified baselines are documented and not simply a present-day sample. The levels of the BCG are biologically recognizable stages in condition of stream water bodies. They can inform a biological basis for biological criteria and regulation of water bodies. Development of quantitative BCG models yield the technical tools for protecting the highest quality waters through TALU and for developing realistic restoration goals for waters affected by legacy activities (e.g., ditching, impoundments). The BCG allows practical and operational implementation of ALUs in a state's water-quality criteria and standards.

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CALIBRATION OF THE BIOLOGICAL CONDITION GRADIENT FOR STREAMS OF MINNESOTA

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EXECUTIVE SUMMARY

The objective of the Clean Water Act is to "restore and maintain physical, chemical, and biological integrity of the Nation's waters." To meet this goal, we need a uniform interpretation of biological condition and operational definitions that are independent of different assessment methodologies. These definitions must be specific, well-defined, and allow for waters of different natural quality and different desired uses. The USEPA has outlined a tiered system of aquatic life use designation, along a gradient (the Biological Condition Gradient, or BCG) that describes how ecological attributes change in response to increasing levels of human disturbance. The BCG is a conceptual model that describes changes in aquatic communities. It is consistent with ecological theory and has been verified by aquatic biologists throughout the United States.

Specifically, the BCG describes how 10 biological attributes of natural aquatic systems change in response to increasing pollution and disturbance. The 10 attributes are in principle measurable, although several are not commonly measured in monitoring programs. The attributes are:

I. Historically documented, sensitive, long-lived, or regionally endemic taxa
II. Sensitive and rare taxa
III. Sensitive but ubiquitous taxa
IV. Taxa of intermediate tolerance
V. Tolerant taxa
VI. Non-native taxa
VII. Organism condition
VIII. Ecosystem functions
IX. Spatial and temporal extent of detrimental effects
X. Ecosystem connectance

The gradient represented by the BCG has been divided into 6 BCG levels of condition that biologists think can be readily discerned in most areas of North America:

1. Natural or native condition
2. Minimal changes in structure of the biotic assemblage and minimal changes in ecosystem function
3. Evident changes in structure of the biotic assemblage and minimal changes in ecosystem function
4. Moderate changes in structure of the biotic assemblage with minimal changes in ecosystem function
5. Major changes in structure of the biotic assemblage and moderate changes in ecosystem function
6. Severe changes in structure of the biotic assemblage and major loss of ecosystem function

This report communicates the development of a quantitative BCG model, consistent with the conceptual model of the BCG of Davies and Jackson (2006). A panel of aquatic biologists in Minnesota applied and calibrated the general BCG model to Minnesota streams. Data from

Minnesota's monitoring program were examined to determine whether the data were adequate to apply to the BCG. The panel was able to assign taxa in the database to the first six attributes listed above, and the panel assigned a set of test sites to BCG levels 1 to 6 based on the sample data.

The panel assigned 728 samples to levels of the BCG—351 benthic macroinvertebrate samples, and 377 fish samples. For some samples, the panel's evaluation reflected some ambiguity between adjacent levels, such that a sample may have had characteristics intermediate between two levels. Level assignments were made across 9 stream types for both fish and benthic macroinvertebrates, including southern coldwater, and northern coldwater streams.

From the panelists' descriptions of their decision criteria for assessing sites and assigning levels, we developed a set of quantitative operational rules for assigning sites to levels. The rules capture the consensus professional judgment of the panel, and can ensure consistent decision-making. The panel's assessments, and the rules, were consistent but not identical across stream classes. The rules were incorporated into a multiple attribute decision model that makes use of mathematical fuzzy-set theory to account for discontinuities and to identify when BCG level assignments may be intermediate between adjacent levels. The purpose of the BCG model is to replicate panel decisions, using the panel-derived rules, so that stream assessments can be automated. The model was incorporated into a stand-alone Microsoft Access application, delivered separately to MPCA. The automated model exactly matched 81% of the fish panel decisions, and 88% of the benthic macroinvertebrate panel decisions. All mismatches were within a single BCG level, and several apparent mismatches were instances where the model, the panel, or both identified a tie between adjacent BCG levels.

The decision rules are documented, so that they have a degree of transparency not available in other index methods (e.g., arithmetic averaging of metrics, development of multivariate discriminant models to identify "true" reference). This also means that the decision rules can be formally changed by future panels as improved information becomes available. The BCG model is appropriate and consistent for use in Minnesota's Tiered Aquatic Life Use development, although we make the following recommendations for strengthening the index over time:

- Test rules with new (unassessed) sites to determine model and panel concordance. Expansion of the calibration dataset could be used to further refine the BCG models and can also help to identify stream reaches that do not fit into the current stream classification framework.
- The BCG rules were more troublesome to develop and readjust to the two headwaters categories: Northern Headwaters and Southern Headwaters. The final BCG models developed for these classes reasonably predicted panel decisions (77-88%), but we recommend that the fish BCG for the two headwater stream classes be reviewed further to demonstrate that the BCG models are consistent.

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Invertebrate Workgroup

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1.0 INTRODUCTION

This report describes the calibration of the Biological Condition Gradient (BCG) to streams of Minnesota. This report translates the conceptual BCG framework into a BCG-based assessment index for use by Minnesota PCA. It can also be applied by sister agencies, and tribes in the ecoregions of Minnesota. The index is calibrated for biological assessment of warm, cold and cold-cool streams of Minnesota. The tool is calibrated for both macroinvertebrate and fish assemblages.

For over a decade, the Minnesota Pollution Control Agency (MPCA) has been using fish and benthic macroinvertebrate assemblage data to assess water resource quality. Until recently, biological indexes have been developed in Minnesota for individual drainage basins. Fish IBI's have been developed for streams in the Minnesota, Red, St. Croix, and Upper Mississippi River Basins (Niemela et al. 1998; Niemela and Feist 2000; 2002), and macroinvertebrate indexes have been developed for the St. Croix and Upper Mississippi basins (Chirhart 2003; Genet and Chirhart 2004). MPCA is currently developing a statewide fish IBI, following the approach in Whittier et al. (2007). The BCG calibration described here relies heavily on the knowledge and experience gained from the previous basin efforts, but it is now intended to be statewide, and addresses MPCA's objective to develop biological criteria for all streams within Minnesota.

The USEPA has supported efforts to develop uniform assessments of aquatic resource condition and to set more uniform aquatic life protection and restoration goals (Davies and Jackson, 2006). These efforts have led to a conceptual model—the BCG— that describes ecological changes, from pristine to completely degraded, that take place in flowing waters with increased anthropogenic degradation (Davies and Jackson, 2006). The BCG framework supports development of biological criteria in a state's water quality standards that can protect the best quality waters; that can be used as a tool to prevent or remediate cumulative, incremental degradation; and that can help to establish realistic management goals for impaired waters. The basis of the framework is recognition that biological condition of waterbodies responds to human-caused disturbance and stress, and that the biological condition can be measured reliably.

This report includes the results of two separate calibration efforts: one to calibrate the BCG for warmwater streams of Minnesota (this report), and the second to calibrate the BCG for cold and cool-water streams in Minnesota, Wisconsin, Michigan, and tribal lands of the region (Gerritsen and Stamp 2013. Results from the multistate calibration are also included here so that all BCG models for Minnesota are in one place.

1.1 What Is the BCG?

Over the past 40 years, states have independently developed technical approaches to assess biological condition and set designated ALUs for their waters. The BCG was designed to provide a means to map different indicators on a common scale of biological condition to facilitate comparisons between programs and across jurisdictional boundaries in context of the CWA. The BCG is a conceptual, narrative model that describes how biological attributes of aquatic ecosystems change along a gradient of increasing anthropogenic stress. It provides a framework for understanding current conditions relative to natural, undisturbed conditions (Figure 1).

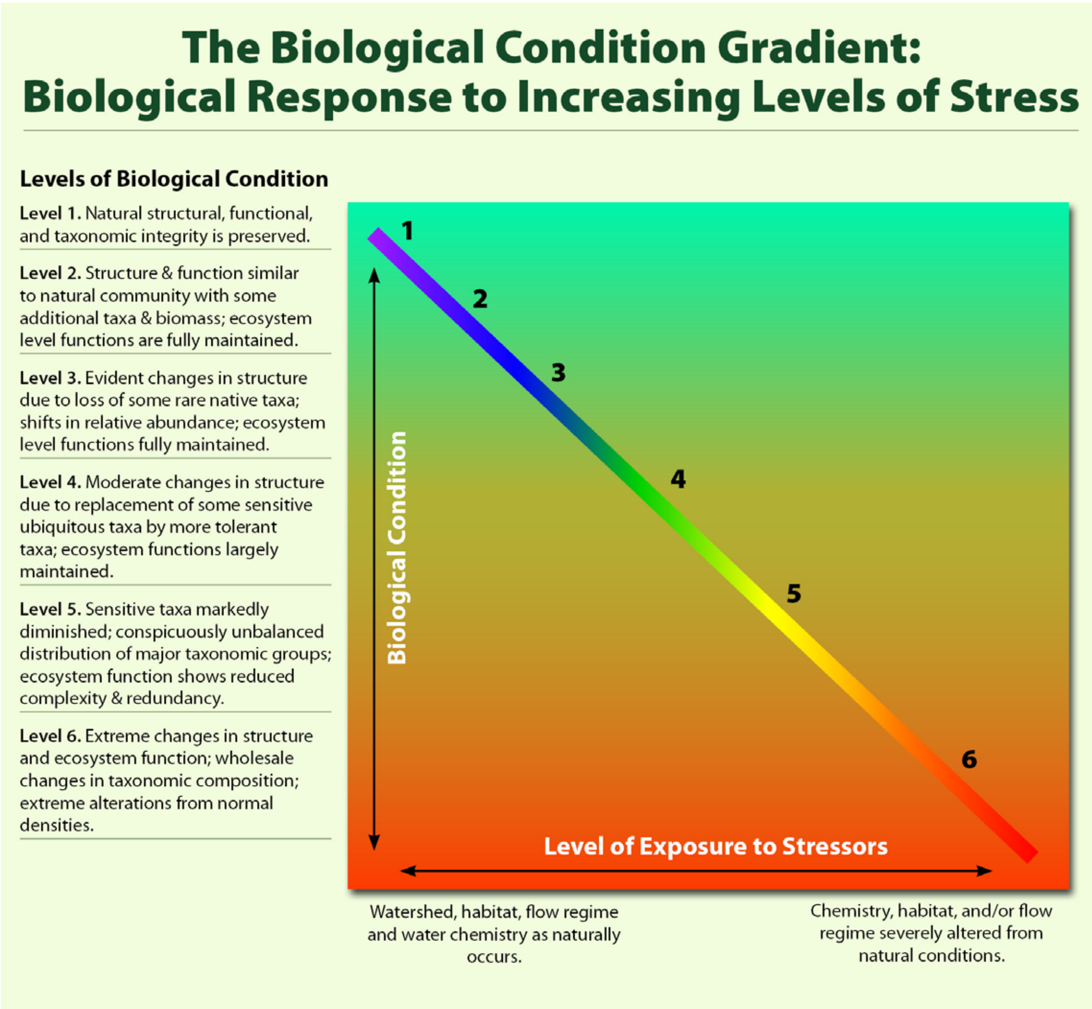

Figure 1. The Biological Condition Gradient (BCG). The BCG was developed to serve as a scientific framework to synthesize expert knowledge with empirical observations and develop testable hypotheses on the response of aquatic biota to increasing levels of stress. It is intended to help support more consistent interpretations of the response of aquatic biota to stressors and to clearly communicate this information to the public, and it is being evaluated and piloted in several regions and states.

*Source: Modified from Davies and Jackson 2006.

The BCG, as a conceptual model, is a universal framework that defines biologically recognizable categories of condition, and the framework is applicable for all states and broad regions. The BCG is not a management system, nor does it describe management goals. However, the reverse is true: management goals can be described in terms of the BCG, and biological information as measured by the BCG can tell us whether criteria are being met. Minnesota can thus identify management goals and levels of protection in terms of the BCG. The highest levels of the BCG could correspond to exceptional natural resource waters, as well as levels to be maintained under antidegradation policy. The interim goal of the Clean Water Act (CWA) (minimally fishable-swimmable) could correspond to the no-longer-pristine middle levels of the BCG, and lower levels would be nonattaining.

A BCG requires strong scientific knowledge on the response of aquatic biological assemblages to stressors, as well as the biota inhabiting a region. Using the scientific information to better assess and manage living aquatic resources also requires a legal foundation that permits the determination of scientifically defensible management goals (policies, designated uses, standards, criteria) in keeping with the goals of the CWA. Finally, developing a quantitative methodology for assessing waterbodies in relation to the BCG requires a scientifically sound biological monitoring program.

Under the CWA a state can identify use classes called designated uses, for its waterbodies. As biological condition can be divided into levels, so can designated aquatic life uses (ALUs) of waterbodies be divided into tiers corresponding to the biological expectation for the different uses. The relationship between ALU tiers and BCG levels must be addressed in the context of the state's programs and policies. BCG development may be required for each tier of ALU (where the ALU tier is defined by environmental classification), or BCG levels may coincide with ALU tiers (where the expected biological condition is the basis for the ALU tier). In this report, we focus on the BCG level development.

Biologists from across the United States developed the BCG model, agreeing that a similar sequence of biological alterations occurs in streams and rivers in response to increasing stress, even in different geographic and climatological areas (Davies and Jackson 2006). The model shows an ecologically-based relationship between the stressors affecting a waterbody (e.g., physical, chemical, biological impacts) and the response of the aquatic community (i.e., biological condition). The model is consistent with ecological theory and can be adapted or calibrated to reflect specific geographic regions and waterbody type (e.g., streams, rivers, wetlands, estuaries, lakes).

The BCG is divided into six levels of biological conditions along the stressor-response curve, ranging from observable biological conditions found at no or low levels of stress (Level 1) to those found at high levels of stress (level 6) (Figure 1). Table 1 provides a description of 10 attributes of aquatic ecosystems that change in response to increasing levels of stressors along the gradient, from level 1 to 6. The attributes include several aspects of community structure, organism condition, ecosystem function, spatial and temporal attributes of stream size, and connectivity. Levels of the condition gradient (Figure 1) are described in greater detail in the following text:

Level 1: Natural or native condition.
Native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability.

> Level 1 represents biological conditions as they existed (or still exist) in the absence of measurable effects of stressors. The Level 1 biological assemblages that occur in a given biogeophysical setting are the result of adaptive evolutionary processes and biogeography that selects in favor of survival of the observed species. For this reason, the expected Level 1 assemblage of a stream from the arid southwest will be very different from that of a stream in the northern temperate forest. The maintenance of native species populations and the expected natural diversity of species are essential for Levels 1 and 2. Non-native taxa (Attribute VI) may be present in Level 1 if they

cause no displacement of native taxa, although the practical uncertainties of this provision are acknowledged.

Attributes I and II (e.g., historically documented and highly sensitive taxa) can be used to help assess the status of native taxa and could be a surrogate measure to identify threatened or endangered species when classifying a site or assessing its condition.

Level 2: Minimal changes in structure of the biotic community and minimal changes in ecosystem function.
Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability.

Level 2 represents the earliest changes in densities, species composition, and biomass that occur as a result of slight elevation in stressors (such as increased temperature regime or nutrient enrichment). There may be some reduction of a small fraction of highly sensitive or specialized taxa (Attribute II) or loss of endemic or rare taxa (Attribute I) as a result. Condition level 2 can be characterized as the first change in condition from natural and it may be manifested in nutrient enriched waters as slightly *increased* richness and density of intermediate sensitive taxa and taxa of intermediate tolerance (Attributes III and IV).

Level 3: Evident changes in structure of the biotic community and minimal changes in ecosystem function.
Evident changes in structure due to loss of some highly sensitive native taxa; shifts in relative abundance of taxa but sensitive-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system.

Level 3 represents readily observable changes that, for example, can occur in response to organic enrichment or increased temperature. The “evident” change in structure for Level 3 is interpreted to be perceptible and detectable decreases in highly sensitive taxa (Attribute II) and increases in opportunist, intermediate tolerant organisms (Attribute IV). Attribute IV taxa (intermediate tolerants) may increase in abundance as an opportunistic response to nutrient inputs.

Level 4: Moderate changes in structure of the biotic community with minimal changes in ecosystem function.
Moderate changes in structure due to replacement of some intermediate-sensitive taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes.

Moderate changes of structure occur as stressor effects increase in Level 4. A substantial reduction of the two sensitive attribute groups (II and III) and replacement by more tolerant taxa (Attributes IV and V) may be observed. A key consideration is that some Attribute III sensitive taxa are maintained at a reduced level but are still an important functional part of the system (function maintained).

Level 5: Major changes in structure of the biotic community and moderate changes in ecosystem function.
Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from those expected; organism condition shows signs of physiological stress; ecosystem function shows reduced complexity and redundancy; increased build-up or export of unused materials.

Changes in ecosystem function (as indicated by marked changes in food-web structure and guilds) are critical in distinguishing between Levels 4 and 5. This could include the loss of functionally important sensitive taxa and keystone taxa (Attribute I, II and III taxa) such that they are no longer important players in the system, though a few individuals may be present. Keystone taxa control species composition and trophic interactions, and are often, but not always, top predators. Additionally, tolerant non-native taxa (Attribute VI) may dominate some assemblages and changes in organism condition (Attribute VII) may include significantly increased mortality, depressed fecundity, and/or increased frequency of lesions, tumors and deformities.

Level 6: Severe changes in structure of the biotic community and major loss of ecosystem function.
Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered.

Level 6 systems are taxonomically depauperate (low diversity and/or reduced number of organisms) compared to the other levels. For example, extremely high or low densities of organisms caused by excessive organic enrichment or severe toxicity may characterize Level 6 systems.

In practice, the BCG is used to first identify the critical attributes of an aquatic community (Table 1-1) and then to describe how each attribute changes in response to stress. Practitioners can use the BCG to interpret biological condition along a standardized gradient, regardless of assessment method, and apply that information to different state or tribal programs.

The BCG model provides a framework to help water quality managers do the following:

- Decide what environmental conditions are desired (goal-setting)—The BCG can provide a framework for organizing data and information and for setting achievable goals for waterbodies relative to “natural” conditions (e.g., condition comparable or close to undisturbed or minimally disturbed condition).
- Interpret the environmental conditions that exist (monitoring and assessment)—Practitioners can get a more accurate picture of current waterbody conditions.
- Plan for how to achieve the desired conditions and measure effectiveness of restoration—The BCG framework offers water program managers a way to help evaluate the effects of stressors on a waterbody, select management measures by which to alleviate those stresses, and measure the effectiveness of management actions.
- Communicate with stakeholders—When biological and stress information is presented in this framework, it is easier for the public to understand the status of the aquatic resources relative to what high-quality places exist and what might have been lost.

Table 1. Attributes used to characterize the BCG.

Attribute	Description
I. Historically documented, sensitive, long-lived, or regionally endemic taxa	Taxa known to have been supported according to historical, museum, or archeological records, or taxa with restricted distribution (occurring only in a locale as opposed to a region), often due to unique life history requirements (e.g., sturgeon, American eel, pupfish, unionid mussel species).
II. Highly sensitive (typically uncommon) taxa	Taxa that are highly sensitive to pollution or anthropogenic disturbance. Tend to occur in low numbers, and many taxa are specialists for habitats and food type. These are the first to disappear with disturbance or pollution (e.g., most stoneflies, brook trout [in the east], brook lamprey).
III. Intermediate sensitive and common taxa	Common taxa that are ubiquitous and abundant in relatively undisturbed conditions but are sensitive to anthropogenic disturbance/pollution. They have a broader range of tolerance than attribute II taxa and can be found at reduced density and richness in moderately disturbed sites (e.g., many mayflies, many darter fish species).
IV. Taxa of intermediate tolerance	Ubiquitous and common taxa that can be found under almost any conditions, from undisturbed to highly stressed sites. They are broadly tolerant but often decline under extreme conditions (e.g., filter-feeding caddisflies, many midges, many minnow species).
V. Highly tolerant taxa	Taxa that typically are uncommon and of low abundance in undisturbed conditions but that increase in abundance in disturbed sites. Opportunistic species able to exploit resources in disturbed sites. These are the last survivors (e.g., tubificid worms, black bullhead).
VI. Nonnative or intentionally introduced species	Any species not native to the ecosystem (e.g., Asiatic clam, zebra mussel, carp, European brown trout). Additionally, there are many fish native to one part of North America that have been introduced elsewhere.
VII. Organism condition	Anomalies of the organisms; indicators of individual health (e.g., deformities, lesions, tumors).
VIII. Ecosystem function	Processes performed by ecosystems, including primary and secondary production; respiration; nutrient cycling; decomposition; their proportion/dominance; and what components of the system carry the dominant functions. For example, shift of lakes and estuaries to phytoplankton production and microbial decomposition under disturbance and eutrophication.
IX. Spatial and temporal extent of detrimental effects	The spatial and temporal extent of cumulative adverse effects of stressors; for example, groundwater pumping in Kansas resulting in change in fish composition from fluvial dependent to sunfish.
X. Ecosystem connectance	Access or linkage (in space/time) to materials, locations, and conditions required for maintenance of interacting populations of aquatic life; the opposite of fragmentation. For example, levees restrict connections between flowing water and floodplain nutrient sinks (disrupt function); dams impede fish migration, spawning.

*Source: Modified from Davies and Jackson 2006.

2.0 METHODS AND DATA

2.1 Developing and Calibrating a Quantitative BCG Model

The BCG defines the response of aquatic biota to increasing levels of stress in a specific region. Although the BCG was developed primarily using forested stream ecosystems, the model can be applied to any region or waterbody by calibrating it to local conditions using specific expertise and local data. To date, many states and tribes are calibrating BCG-based indexes using the first seven attributes (Table 1) that characterize the biotic community primarily tolerance to stressors, presence/absence of native and nonnative species, and organism condition.

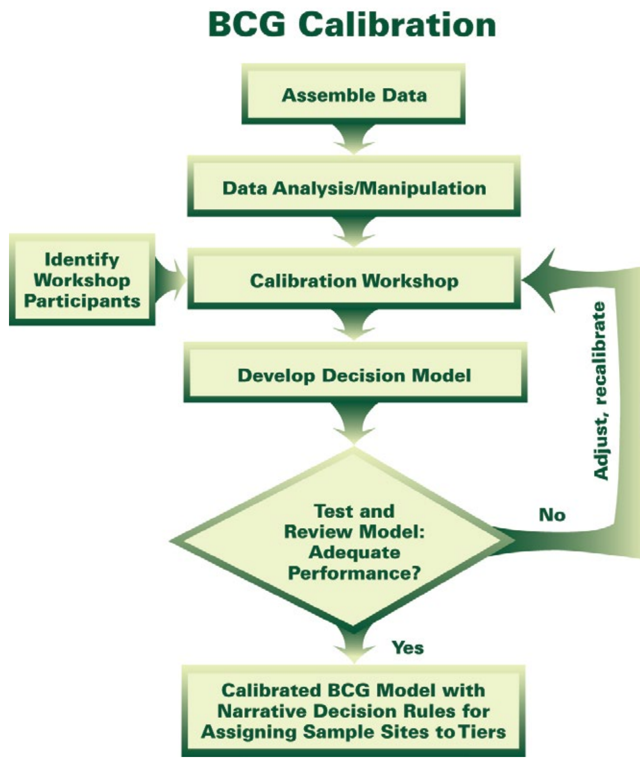

Figure 2. Steps in a BCG calibration.

Calibrating a BCG model to local conditions (Figure 2) is a multistep process. The process is followed to describe the native aquatic assemblages under natural conditions; identify the predominant regional stressors; and describe the BCG, including the theoretical foundation and observed assemblage response to stressors. Index calibration begins with the assembly and analysis of biological monitoring data. A calibration workshop is held at which experts familiar with local conditions use the data to define the ecological attributes and set narrative statements. For example, the experts determine narrative decision rules for assigning sites to a BCG level on the basis of the biological information collected at sites. Documentation of expert opinion in assigning sites to tiers is a critical part of the process. A decision model is then developed that encompasses those rules and is tested with independent data sets. A decision model based on the tested decision rules is a transparent, formal, and testable method for documenting and validating expert knowledge. A quantitative data analysis program can then be developed using those rules.

2.1.1 Assigning Sites to BCG Levels

Aquatic biologists familiar with Minnesota streams met as a workgroup to develop both the ecological attributes and rules for assigning sites to levels in the gradient. Their expertise included aquatic ecology, benthic macroinvertebrate sampling and monitoring, water quality, and fisheries biology. This development of the gradient included systematic application to both benthic macroinvertebrates and fish, collected by the methods used in Minnesota's monitoring program. As in other applications, we developed the BCG using only attributes I–VI, because the monitoring program does not collect information on the other attributes.

After reviewing EPA's conceptual model of the BCG, the group reviewed the list of taxa identified in the Minnesota ambient monitoring program to assign taxa to attribute groups I–VI. Appendix A includes the taxa list and assigned attribute groups. The group then considered data from selected monitoring sites, and assigned the sites to levels of the BCG based on the taxa present in the sample.

The conceptual model of the BCG is universal (Davies and Jackson 2006; USEPA 2005), but descriptions of communities, species, and their responses to the stressor gradient are specific to the conditions and communities found in the sample region. The expert panel described the biological condition levels that can be discerned within Minnesota. The description of natural conditions requires biological knowledge of the region, a natural classification of the assemblages, and, if available, historical descriptions of the habitats and assemblages. Working from the description of undisturbed communities and species composition data from example sites, the panel then assigned sites to the levels of the BCG. These site assignments were used to describe changes in the aquatic communities for lower levels of biological condition, leading to a complete descriptive model of the BCG for the region. Throughout this process, the panel made use of the prepared data, examining species composition and abundance data from sites with different levels of cumulative stress, from least stressed to severely stressed. Samples were selected by data analysts; the panel was initially unaware of the stressor status of individual sites. The panel worked with data tables showing the species and attributes for each site. In developing assessments, the panel worked "blind", that is, no stressor information was included in the data table. Only non-anthropogenic classification variables were shown. Panel members discussed the species composition and what they expected to see for each level of the BCG, for example, "I expect to see more stonefly taxa in a BCG level 2 site."

2.1.2 Quantitative Description

Level descriptions in the conceptual model tend to be rather general (e.g., "reduced richness"). To allow for consistent assignments of sites to levels, it is necessary to formalize the expert knowledge by codifying level descriptions into a set of rules (e.g., Droesen 1996). If formalized properly, any person (with data) can follow the rules to obtain the same level assignments as the group of experts. This makes the actual decision criteria transparent to stakeholders.

Rules are logic statements that experts use to make their decisions; for example, "If taxon richness is high, then biological condition is high." Rules on attributes can be combined, for example: "If the number of highly sensitive taxa (attribute II) is high, <u>and</u> the number of tolerant individuals (attribute V) is low, then assignment is level 2." In questioning individuals on how decisions are made in assigning sites to levels, people generally do not use inflexible, "crisp" rules, for example, the following rule is unlikely to be adopted:

"Level 2 always has 10 or more attribute II taxa; 9 attribute II taxa is always level 3."

Rather, people use strength of evidence in allowing some deviation from their ideal for any individual attributes, as long as most attributes are in or near the desired range. Clearly, the definitions of "high," "moderate," "low," etc., are fuzzy. These rules preserve the collective professional judgment of the expert group and set the stage for the development of models that reliably assign sites to levels without having to reconvene the same group. In essence, the rules and the models capture the panel's collective decision criteria.

As the panel assigned example sites to BCG levels, the members were polled on the critical information and criteria they used to make their decisions. These formed preliminary, narrative rules that explained how panel members made decisions. For example, "For BCG level 2, sensitive taxa must make up half or more of all taxa in a sample." The decision rule for a single level of the BCG does not always rest on a single attribute (e.g., highly sensitive taxa) but may include other attributes as well (intermediate sensitive taxa, tolerant taxa, indicator species), so these are termed "Multiple Attribute Decision Rules." With data from the sites, the rules can be checked and quantified. Quantification of rules allows users to consistently assess sites according to the same rules used by the expert panel, and allows a computer algorithm, or other persons, to obtain the same level assignments as the panel.

Rule development requires discussion and documentation of BCG level assignment decisions and the reasoning behind the decisions. During this discussion, we recorded:

- Each participant's decision ("vote") for the site
- The critical or most important information for the decision—for example, the number of taxa of a certain attribute, the abundance of an attribute, the presence of indicator taxa, etc.
- Any confounding or conflicting information and how this was resolved for the eventual decision

Following the initial site assignment and rule development, we developed descriptive statistics of the attributes and other biological indicators for each BCG level determined by the panel. These descriptions assisted in review of the rules and their iteration for testing and refinement.

Rule development is iterative, and may require several panel sessions. Following the initial development phase, the draft rules were tested by the panel with new data to ensure that new sites are assessed in the same way. The new test sites were not used in the initial rule development and also should span the range of anthropogenic stress. Any remaining ambiguities and inconsistencies from the first iterations were also resolved.

2.1.3 Decision Criteria Models

Consensus professional judgment used to describe the BCG levels can take into account nonlinear responses, uncommon stressors, masking of responses, and unequal weighting of attributes. This is in contrast to the commonly used biological indexes, which are typically unweighted sums of attributes (e.g., multimetric indexes; Barbour et al. 1999; Karr and Chu 1999), or a single attribute, such as observed to expected taxa (e.g., Simpson and Norris 2000; Wright 2000). Consensus assessments built from the professional judgment of many experts result in a high degree of confidence in the assessments, but the assessments are labor-intensive (several experts must rate each site). It is also not practical to reconvene the same group of experts for every site that is monitored in the long term. Since experts may be replaced on a panel over time, assessments may in turn "drift" due to individual differences of new panelists.

Management and regulation, however, require clear and consistent methods and rules for assessment, which do not change unless deliberately reset.

Use of the BCG in routine monitoring and assessment thus requires a way to automate the consensus expert judgment so that the assessments are consistent. We codified the decision criteria into a decision model, which has the advantage that the criteria are visible and transparent.

Codification of Decision Criteria

The expert rules can be automated in Multiple Attribute Decision Models. These models replicate the decision criteria of the expert panel by assembling the decision rules using logic and set theory, in the same way the experts used the rules. Instead of a statistical prediction of expert judgment, this approach directly and transparently converts the expert consensus to automated site assessment. The method uses modern mathematical set theory and logic (called "fuzzy set theory") applied to rules developed by the group of experts. Fuzzy set theory is directly applicable to environmental assessment, and has been used extensively in engineering applications worldwide (e.g., Demicco and Klir 2004) and environmental applications have been explored in Europe and Asia (e.g., Castella and Speight 1996; Ibelings et al. 2003).

Mathematical fuzzy set theory allows degrees of membership in sets, and degrees of truth in logic, compared to all-or-nothing in classical set theory and logic. Membership of an object in a set is defined by its membership function, a function that varies between 0 and 1. To illustrate, we compare how classical set theory and fuzzy set theory treat the common classification of sediment, where sand is defined as particles less than or equal to 2.0 mm diameter, and gravel is greater than 2.0 mm (Demicco and Klir 2004). In classical "crisp" set theory, a particle with diameter of 1.999 mm is classified as "sand", and one with 2.001 mm diameter is classified as "gravel." In fuzzy set theory, both particles have nearly equal membership (approximately 0.5) in both classes (Demicco 2004). Very small measurement error in particle diameter greatly increases the uncertainty of classification in classical set theory, but not in fuzzy set theory (Demicco and Klir 2004). Demicco and Klir (2004) proposed four reasons why fuzzy sets and fuzzy logic enhance scientific methodology:

- Fuzzy set theory has greater capability to deal with "irreducible measurement uncertainty," as in the sand/gravel example above.
- Fuzzy set theory captures vagueness of linguistic terms, such as "many," "large" or "few."
- Fuzzy set theory and logic can be used to manage complexity and computational costs of control and decision systems.
- Fuzzy set theory enhances the ability to model human reasoning and decision-making, which is critically important for defining thresholds and decision levels for environmental management.

Development of the BCG

In order to develop the fuzzy inference model, each linguistic variable (e.g., "high taxon richness") must be defined quantitatively as a fuzzy set (e.g., Klir 2004). A fuzzy set has a membership function; example membership functions of different classes of taxon richness are shown in Figure 3. In this example (Figure 3), piecewise linear functions (functions consisting of line segments) are used to assign membership of a sample to the fuzzy sets. Numbers below a lower threshold have membership of 0, and numbers above an upper threshold have membership of 1, and membership is a straight line between the lower and upper thresholds. For example, in Figure 3, a sample with 20 taxa would have a membership of approximately 0.5 in the set "low to moderate Taxa" and a membership of 0.5 in the set "Moderate Taxa."

How are inferences made? Suppose there are two rules for determining if a waterbody is BCG Level 3 (using definitions of Figure 2-2):

- The number of total taxa is high

The number of sensitive taxa is low to moderate

In crisp set theory, these rules translate to:

- Total taxa > 27
- Sensitive taxa > 10

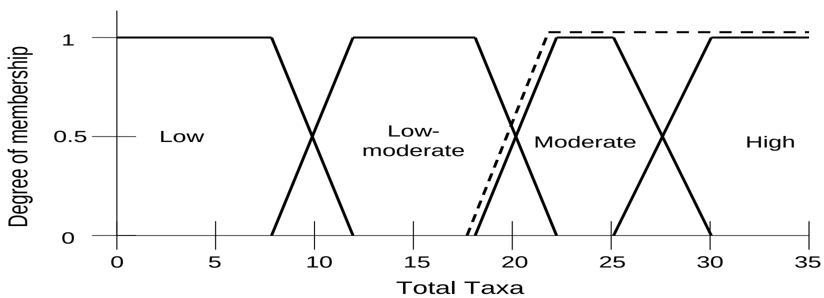

Figure 3. Fuzzy set membership functions assigning linguistic values of Total Taxa to defined quantitative ranges. Heavy dashed line shows membership of fuzzy set defined by "Total taxa are moderate to high."

If the two rules are combined with an "AND" operator, that is, both must be true, then under crisp set theory, if total taxa = 28 and sensitive taxa = 10, the sample would be judged not to be in the set of BCG level 3. This is because sensitive taxa is 1 short of being greater than 10.

In fuzzy set theory, an AND operator is equivalent to the minimum membership given by each rule: level 3 = MIN (total taxa is high, sensitive taxa is low to moderate)

Fuzzy membership in "total taxa is high" = 0.6 (Figure 2-2), and fuzzy membership in "sensitive taxa is low to moderate" = 0.5 (Figure 2-2). Membership of Level 3 is then 0.5

If the two rules are combined with an "OR" operator, then either can be true for a site to meet BCG level 3, and both conditions are not necessary. Crisp set theory now yields a value of "true" if total taxa = 28 and sensitive taxa = 10 (total taxa > 27, therefore it is true). Fuzzy set theory yields a membership of 0.6 (maximum of 0.5. and 0.6). Using the fuzzy set theory model, finding an additional taxon in a sample does not cause the assessment to flip to another class, unlike crisp decision criteria.

2.2 Data—Minnesota's Water Monitoring Program

Consistent, high quality biological monitoring information is necessary for developing a quantitative assessment system within a BCG framework. MPCA operates a sizable ambient monitoring program throughout the state; as of 2011, MPCA had recorded more than 5000 fish sampling events and nearly 3000 macroinvertebrate sampling events in its database.

Sites may be selected for assessment for a number of reasons including: 1) sites randomly selected for condition monitoring as part of the Environmental Monitoring and Assessment Program (EMAP), 2) sites selected for the development and calibration of biological criteria, and 3) sites selected to evaluate a suspected source of pollution.

2.2.1 Fish sampling

Fish collection Standard Operating Procedures (SOPs) were extracted from MPCA (2009), *Fish Community Sampling Protocol for Stream Monitoring Sites*,[1] and are summarized below:

A fish sampling reach is defined as 35 times the mean stream width, and is based on the distance necessary to capture a representative and repeatable sample of the fish assemblage within a stream segment (Lyons 1992; cited in MPCA 2009). Sampling is conducted during daylight hours within the summer index period of mid-June through mid-September. Sampling should occur when streams are at or near base-flow because flood or drought events can have a profound effect on fish assemblage structure and sampling efficiency.

Fish are collected before the physical habitat assessment so as not to disturb the fish assemblage prior to sampling. All habitat types within the sampling reach are sampled in the approximate proportion that they occur. An effort is made to collect all fish observed, but fish < 25 mm in total length are not counted as part of the catch. Fish are collected with electrofishing, using one of 4 methods: backpack shocker in small headwater streams; towed stream shocker in larger wadeable streams; mini-boom shocker (2-person jonboat) in small, non-wadeable streams, and a larger boom shocker (boat mounted) in larger streams and rivers.

2.2.2 Macroinvertebrate sampling

Macroinvertebrate collection SOPs were extracted from MPCA (undated), *Invertebrate Sampling Procedures*, EMAP-SOP4, Rev. 0.[2]

[1] http://www.pca.state.mn.us/index.php/water/water-monitoring-and-reporting/biological-monitoring/stream-monitoring/stream-monitoring-fish.html
[2] http://www.pca.state.mn.us/index.php/water/water-monitoring-and-reporting/biological-monitoring/stream-monitoring/stream-monitoring-aquatic-invertebrates.html

The multihabitat method entails collecting a composite sample from up to five different habitat types to get a sample representative of the invertebrate assemblage of a particular sampling reach. The habitats were chosen to represent broad categories rather than microhabitats. Every broad category includes numerous microhabitats, some of which will not be sampled. Habitats are sampled to reflect the most common microhabitat of any given broad habitat category. The habitats to be sampled include:

- *Hard bottom (riffle/cobble/boulder)*—All hard, rocky substrates, not just riffles. Runs and wadeable pools often have suitable "hard" substrates, and should not be excluded from sampling. Unproductive surfaces of large boulders and areas of flat, exposed bedrock are avoided unless they are productive.

- *Aquatic Macrophytes (submerged/emergent vegetation)*—Any vegetation found at or below the water surface. Emergent vegetation is included because all emergent plants have stems that extend below the water surface, serving as suitable substrate for macroinvertebrates.

- *Undercut Banks (undercut banks/overhanging vegetation)*—This category is meant to cover in-bank or near-bank habitats, shaded areas away from the main channel that typically are buffered from high water velocities.

- *Snags (snags/rootwads)*—Snags include any piece of large woody debris found in the stream channel, and include, rootwads, logs, tree trunks, entire trees, tree branches, large pieces of bark, and dense accumulations of twigs.

- *Leaf Packs*—Leaf packs are dense accumulations of leaves typically present in the early spring and late fall. They are found in deposition zones, generally near stream banks, around logjams, or in current breaks behind large boulders.

Sampling consists of dividing 20 sampling efforts equally among the dominant, productive habitats present in the reach. If 2 habitats are present, each habitat receives 10 sampling efforts. If 3 habitats are present, the two most dominant habitats should receive 7 jabs, the third should receive 6 jabs. If a productive habitat is present in a reach but not in great enough abundance to receive an equal proportion of sampling efforts, it is thoroughly sampled and the remaining samples should be divided among the remaining habitat types present.

A sample effort is defined as taking a single dip or sweep in a common habitat. A sweep is taken by placing the D-net on the substrate and disturbing the area directly in front of the net opening equal to the net width, ca. 1ft^2. The net is swept several times over the same area to ensure that an adequate sample is collected; each sweep covers approximately .09 m^2 of substrate. Total area sampled is ca. 1.8 m^2.

2.2.3 Data Management

Currently, all of MPCA's fish and benthic data and associated metadata are entered into a Microsoft Access database, where metrics and summary information are generated through queries. MPCA provided Tetra Tech with an extract of the database that included more than 5000 valid fish samples and sites and approximately 3000 benthic macroinvertebrate samples for use in the calibration exercise.

2.3 Identifying Attributes

2.3.1 Preliminary Disturbance Gradient

MPCA has developed a disturbance index, based on watershed land use, stream alteration, riparian condition, and known permitted discharges. Disturbance index score can range from 1, representing completely altered and heavily stressed streams, to 81, representing nearly pristine watersheds.

2.3.2 Assignment of taxa to attributes

Biologists have long observed that taxa differ in their sensitivity to pollution and disturbance. While biologists largely agree on the relative sensitivity of taxa, there may be subtle differences among stream types (high vs. low gradient) or among geographic regions. We applied several statistical models to estimate tolerance of fish and macroinvertebrates to stressors, in this case MPCA's disturbance gradient. The workgroup participants examined the empirical information derived from the models, as well as using their collective experience and judgment to assign sensitivities of the organisms to the disturbance gradient.

Quantitative tolerance models
Prior to the workshops, we examined tolerances of the fish and macroinvertebrate taxa to the stressor gradient. While optima or tolerance values can be estimated from a variety of models, scatterplots of individual taxa on the disturbance gradient, and a maximum likelihood model of the probability of observing a taxon at a particular disturbance score were deemed the most useful for assigning taxa to the tolerance attributes.

Maximum likelihood estimates (GLM model)—The probability of observing a particular taxon can be modeled as:

$$\ln\left[\frac{p}{1-p}\right] = b_0 + b_1 x + b_2 x^2$$

Where p is the probability of observing the taxon and x is the disturbance gradient score. The optimum of the model (maximum probability) yields the tolerance value. To assist experts in assigning taxa to attributes, we plotted the probability over the range of the disturbance gradient (See Figure 3-1).

Prior to calibrating BCG levels, the two workgroups (fish and benthic macroinvertebrates) assigned Minnesota taxa to the taxonomic attribute groups (attributes I to VI; Section 1.1.1). Assignments of taxa to attributes relied on a combination of empirical examination of taxon occurrences at sites in the different stress classes, as well as professional experience of field biologists who had sampled the streams of Minnesota. The empirical analyses and professional opinions tended to agree, but in cases of disagreement, the group relied on consensus professional opinion, unless contradicted by an overwhelming response in the data analysis. As a group, participants discussed each taxon in the calibration data set, and developed a consensus assignment (Appendix A).

2.4 Classification

Experience has shown that a robust biological classification is necessary to calibrate a BCG-based index, because the natural biological class indicates the species expected to be found in undisturbed, high-quality sites. As an example, low-gradient prairie or wetland-influenced streams typically contain species that are adapted to slow-moving water and often to hypoxic conditions. These same species found in a high-gradient, forest stream could indicate habitat degradation and organic enrichment.

MPCA had previously developed classification systems for both the fish and the benthic macroinvertebrate communities, with 11 fish classes and 12 macroinvertebrate classes for streams. These classes were based on distributions of species among Minnesota's ecoregions (forest, prairies), a north-south gradient, stream size for fish samples (headwater, wadeable, and river), and stream gradient for macroinvertebrate samples (Riffle-run and Glide-pool).

The first BCG calibration exercise was done on 19 of the above stream classes (excluding 4 coldwater classes), but after the workshop MPCA re-examined the classifications to see if some of the classes could be recombined to reduce the total number of classes. The objective was to reduce the complexity of the assessment system, as well as to ensure a more complete stress/disturbance gradient for each stream class. A revised set of stream classes was developed by MPCA from further data analysis and examination of results from the calibration exercises (Table 2). The final classification identified 7 warmwater stream classes for both fish and benthic macroinvertebrates, and 2 cold and coolwater classes (Table 2-1), for a total of 18 classes, 9 each for fish and invertebrates.

Table 2. Final MPCA classification of stream types for fish and macroinvertebrates, and number of samples with valid data in each (through September 2011).

Fish			Benthic macroinvertebrates		
MPCA no.	Name	N	MPCA no.	Name	N
1	Prairie Rivers	525	1	Northern Forest River	125
2	Southern Wadeable Streams	665	2	Prairie Rivers (north and south)	155
3	Southern Headwaters	638	3	Northern Forest Riffle-run	271
4	Northern Forest Rivers	358	4	Northern Forest Glide-pool	425
5	Northern Wadeable Streams	523	5	Southern Riffle-run	445
6	Northern Headwaters	706	6	Southern Hardwood Glide-pool	396
7	Wetland-lacustrine Streams	313	7	Prairie Glide-pool	617
10	Southern Coldwater	288	8	Northern Coolwater	166
11	Northern Coolwater	628	9	Southern Coldwater	245

3.0 DESCRIPTIVE RESULTS

MPCA hosted workshops and webinars to develop the rules and models for warmwater streams. USEPA hosted additional workshops and webinars for cold- and coolwater streams, for the 3 states and several tribes in northern-most EPA Region 5 (Gerritsen and Stamp 2012). Following the coldwater BCG development, MPCA subsequently refined the cold and coolwater BCG models to obtain better fits to MPCA data.

In the final webinars for both warmwater and coldwater calibration, the panels assessed sites that were not used in the calibration of the BCG model, to serve as independent tests of model performance. Several of these sites were used for MPCA's final refinement of the index models, so they can no longer be considered independent test sites for the current configuration of the models.

In this process, panelists first assigned BCG attributes to fish and macroinvertebrate taxa (See section 3.1). Next they examined biological data from individual sites and assigned those samples to Levels 1 to 6 of the BCG. The intent was to achieve consensus and to identify rules that experts were using to make their assignments. Panelists operated on the assumption that sites had been classified correctly into the stream types identified in Table 2.

The data that the experts examined when making BCG level assignments were provided in worksheets. The worksheets contained lists of taxa, taxa abundances, BCG attribute levels assigned to the taxa, BCG attribute metrics and limited site information (e.g., such as watershed area), size class (i.e., headwater), and stream gradient. Participants were not allowed to view Station IDs or waterbody names when making BCG level assignments, as this might bias their assignments. Fish and macroinvertebrate worksheets can be found in Appendix D.

Preliminary sets of decision rules were developed based on these calibration worksheets. The rules were automated in Excel spreadsheets and BCG level assignments were calculated for each sample. The model-assigned BCG level assignments were then compared to the BCG level assignments that had been made by the panelists to evaluate model performance. A second workshop and several webinars were held to reconsider samples that had the greatest differences between the BCG level assignments based on the model versus the panelists. Decision rules were adjusted based on group consensus. After the decision rules were finalized, Tetra Tech also developed an application in MS-Access for automated calculation of BCG level for new sample data.

3.1 BCG Taxa Attributes

Scatterplots of abundance of individual taxa on the disturbance gradient, which also showed the maximum likelihood model, was deemed to be the most useful for identifying attribute groups (Figure 4). Scatterplots were plotted for all taxa with more than 20 occurrences in the data set (Appendix B). Figures 4-7 show examples of the scatterplots and maximum likelihood models for taxa assigned to attributes II through V. Undisturbed sites score high on the Minnesota disturbance gradient (maximum score = 81). The scatterplots of relative abundance (points shown in Figs. 4-7) may be misleading because the distribution of the disturbance scores is not uniform: there are many more sites in the database with scores above 40 than scores below 40. An apparent reduction in point density at low disturbance scores reflects the fact that few sites in the database had such low scores, and not necessarily the response of the taxa. The capture probability curve shows better which taxa are most tolerant to, or indeed thrive in, disturbed conditions (Figure 4).

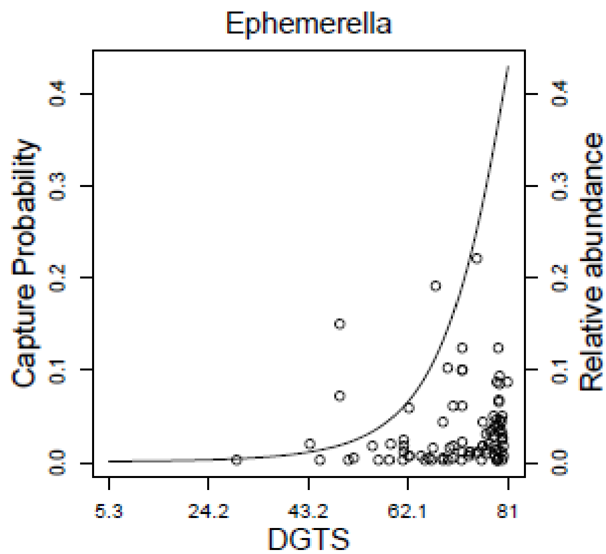

Figure 4. Disturbance score and Ephemerella occurrence in stream samples. Circles show observations and relative abundance of Ephemerella (right axis); line shows probability of occurrence (left axis; maximum likelihood). Ephemerella was assigned to attribute II (highly sensitive taxa), as shown by its high abundance and high probability of occurrence in minimally-disturbed sites (disturbance score 81).

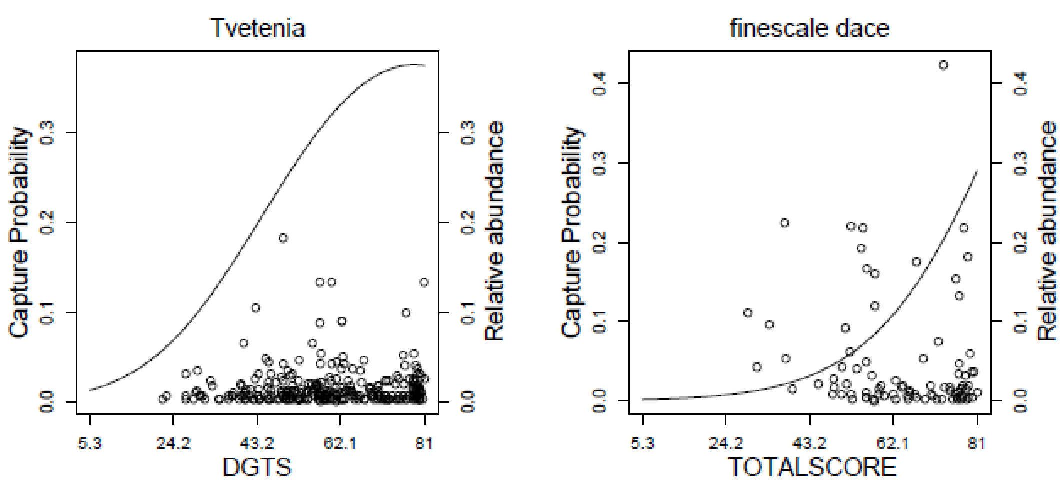

Figure 5. Examples of attribute III taxa, *Tvetenia* and finescale dace. These species occur throughout the disturbance gradient, but with higher probability in better sites. Final attribute assignment was based not only on these plots, but also on professional judgment of the panel.

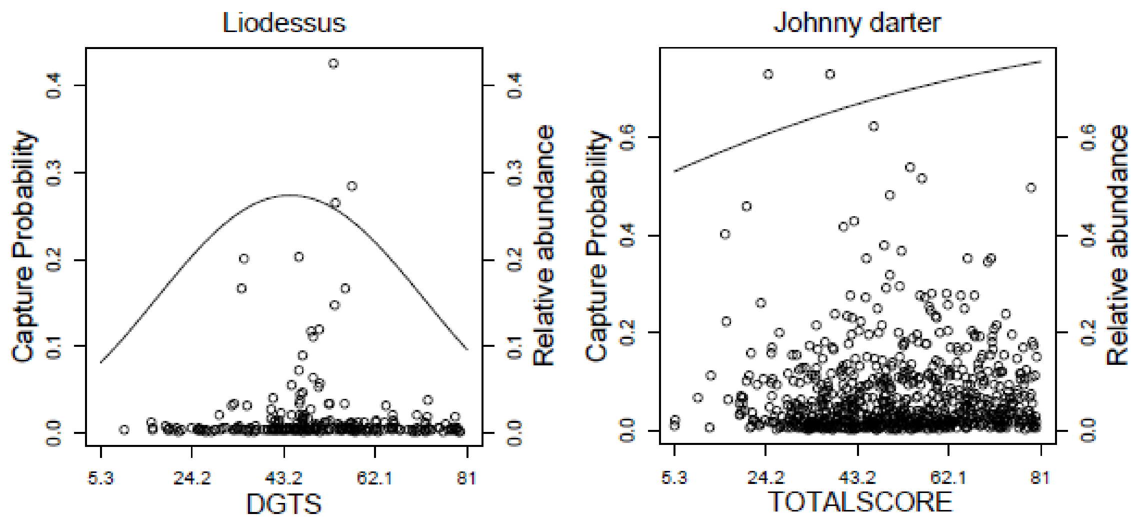

Figure 6. Examples of intermediate tolerant, attribute IV taxa, *Liodessus* and johnny darter. These species occur throughout the disturbance gradient, but with roughly equal probability throughout, or with a peak in the middle of the disturbance range.

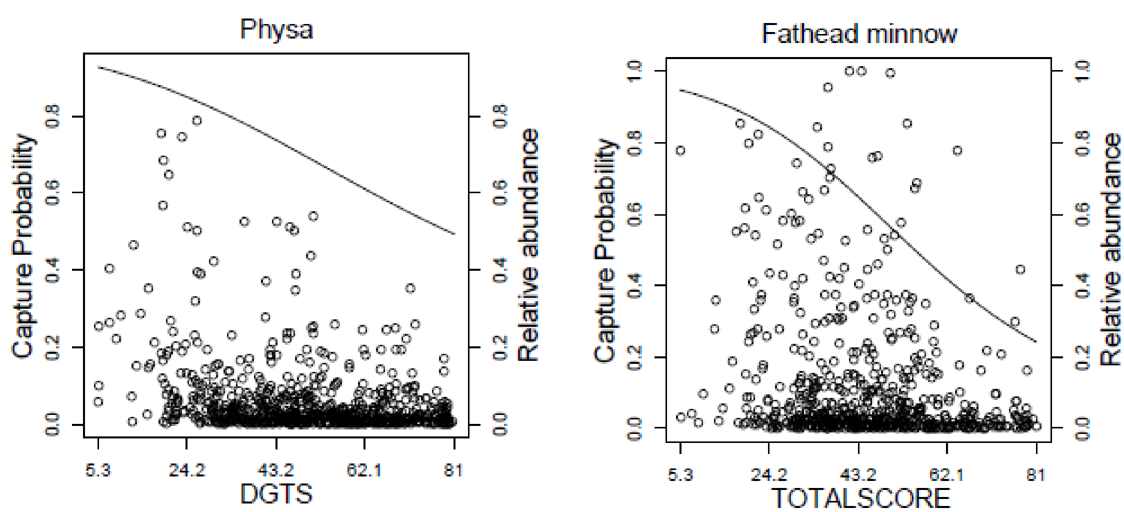

Figure 7. Examples of tolerant (or highly tolerant) attribute V taxa, *Physa* and fathead minnow (Va; highly tolerant). These species occur throughout the disturbance gradient, but with higher probability of occurrence, and higher abundances, in more stressed sites.

Fish species were assigned to attributes separately for each of the 9 fish stream classes, and macroinvertebrates were assigned separately to 4 classes: glide-pool, riffle-run, coolwater, and coldwater. One or more taxa differed in attribute assignment in each of the stream classes, although the majority of taxa were in the same attribute among most classes where they occurred.

To illustrate different tolerance among the stream classes, we show the tolerance graphics for creek chub, compared in the wadeable streams and Headwaters classes (Figure 8). Based on the graphics, creek chub appears to be more tolerant in the wadeable streams than in headwaters.

Other species (e.g., fathead minnow, attribute V) appeared the same in both wadeable and headwaters. Attribute assignments for all taxa among the stream classes are given in Appendix A.

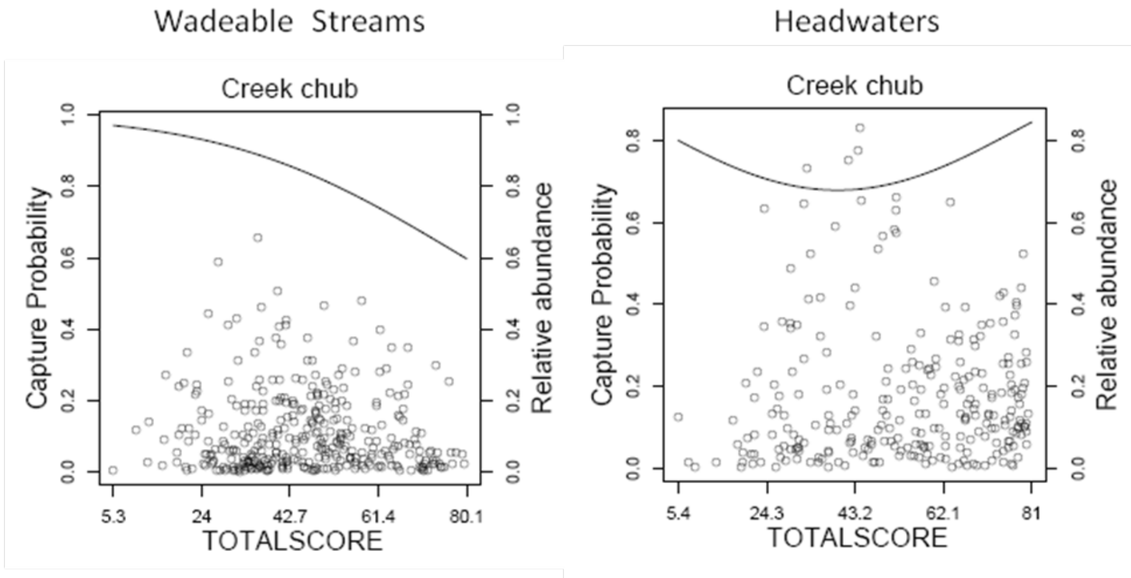

Figure 8. Tolerance graphics for creek chub in wadeable streams (left) and headwaters (right). In wadeable streams, creek chub is tolerant, an attribute V species. In headwaters, Creek chub appears equally likely to occur in the nearly all sites, making it a species of intermediate tolerance (attribute IV).

Fish experts identified two additional attributes of highly tolerant taxa, the most tolerant fishes (attribute V-a; the last survivors in the most highly stressed sites, and further divided the nonnative into moderately sensitive nonnative salmonids (Attribute VI; including brown trout and rainbow trout); and highly tolerant non-salmonid, nonnative species (attribute VI-a, including ruffe, sea lamprey, carp). The distinction separating the highly tolerant attribute V-a fish from the merely tolerant attribute V was based on the collective professional experience and judgment of the fish panel. The panel was of the opinion that identifying the highly tolerant V-a and VI-a attributes would improve discrimination of BCG levels.

A summary breakdown of taxa by attribute group is shown in Tables 3 and 4. The Minnesota taxa lists and final attribute assignments are given in Appendix A. More than 100 invertebrate taxa were left unassigned because participants felt there was insufficient information on the taxa, or they were relatively unusual in the data set. Only 2 fish were left unclassified; both hybrids.

Table 3. Examples of macroinvertebrate taxa by attribute group. Assignment to attribute varied for some taxa among habitat (glide-pool and riffle-run), and stream temperature class (warmwater and cold-cool).

Ecological Attribute	Number of genera*	Example Taxa
I Endemic, rare	1-2	*Goera, Apatania* (cold and cool only)
II Highly Sensitive	29-41	*Stempellina, Heleniella, Ephemerella, Paraleuctra, Ophiogomphus, Parapsyche, Diplectrona, Lepidostoma, Dolophilodes, Rhyacophila*
III Intermediate Sensitive	107-148	*Diamesa, Tvetenia, Hexatoma, Plauditus, Parapoynx, Isoperla, Boyeria, Amphinemura, Pycnopsyche, Brachycentrus, Limnephilus*
IV Intermediate Tolerant	201-231	Dytiscidae, Ceratopogonidae, *Polypedilum, Limonia, Perlesta, Heptagenia, Libellula, Hydropsyche, Sphaerium, Planorbella*
V Tolerant	25-41	Erpobdellidae, *Cricotopus, Pseudocloeon,* Corixidae, *Enallagma, Caecidotea,* Physidae
VI Nonnative	1	*Corbicula*
x Unassigned	33	Family identifications or unusual taxa; *Chaoborus, Zavrelia, Didymops*, Nemata

* range of number of genera assigned to attribute group among 4 groups

Table 4. Examples of fish taxa by attribute group.

Ecological Attribute	Number of species*	Example Species
I Endemic, rare	1 - 9	blue sucker, crystal darter, gilt darter, greater redhorse, lake sturgeon, pugnose shiner, river redhorse, shovelnose sturgeon, Topeka shiner
II Highly Sensitive	6 - 17	American brook lamprey, blackchin shiner, brook trout, southern brook lamprey, western sand darter
III Intermediate Sensitive	15 - 35	blacknose shiner, burbot, golden redhorse, hornyhead chub, shorthead redhorse, smallmouth bass
IV Intermediate Tolerant	26 - 43	common shiner, gizzard shad, johnny darter, northern pike, spotfin shiner, white sucker[1]
V Tolerant	5 - 18	creek chub, brassy minnow, brook stickleback, central stoneroller, sand shiner
V-a Highly tolerant	7 - 8	bigmouth shiner, bluntnose minnow, fathead minnow, green sunfish
VI Sensitive Nonnative	3	brown trout, rainbow trout, chinook salmon
VI-a Tolerant nonnative	4	common carp, goldfish, ruffe, threespine stickleback
x unassigned		Unidentified fish, hybrids

*Range of numbers of species assigned to attribute among 9 stream types.
[1] White sucker is classed "tolerant" (attribute V) in wadeable streams only

3.2 Site Assignments to BCG Levels

The workgroup examined macroinvertebrate data from 351 samples (9 stream classes), and fish data from 377 samples (9 stream classes). The group was able to reach a majority opinion on the BCG level assignments for all sites reviewed. Data files used in the workshops are in Appendix D, and are summarized in Appendix C. In some cases, there was discussion and some disagreement on which of two adjacent BCG levels a site should be assigned to. These sites were apparently intermediate, with characteristics of both of the adjacent BCG levels.

The panels were able to distinguish 6 separate BCG levels (BCG Levels 1-6), although both levels 1 (nearly pristine) and 6 (extreme degradation) were rare. Nine level 1 samples were identified by the fish group (Appendix C, D), but none were identified by the macroinvertebrate group. In general, macroinvertebrate experts felt that Level 1 and Level 2 sites are not distinguishable using macroinvertebrate data only, in part because rare and endemic taxa are poorly identified, their historic distributions are very poorly known, and finally, the macroinvertebrate sampling methodology is extremely inefficient at finding rare and endemic species. Further examination may be necessary to determine if these sites meet criteria for “minimally disturbed” (Stoddard et al., 2006). Nine level 6 samples were identified by the macroinvertebrate group, and eight by the fish group.

3.3 Attributes and BCG Levels

Examinations of taxonomic attributes among the BCG levels determined by the panels showed that several of the attributes are useful in distinguishing levels, and indeed, were used by the biologists for decision criteria. We derived metrics relating to the attributes (taxa richness, percent of taxa, percent of individuals, dominance, etc.). Metric values, by BCG level, are graphically presented as box and whisker plots in Figures 9-16, and statistical summaries of each metric and BCG level are given in Appendix C.

Several generalizations can be made from the panel’s assignments:

Warmwater invertebrates (Figures 9-11):

- Total taxa richness declines from BCG level 2 to poorer BCG levels, but there is much overlap between adjacent BCG Levels.
- Attribute I and II taxa occur in BCG level 2, but decline markedly in Level 3, and are generally absent in levels 4-6
- All sensitive taxa (attributes I, II, and III combined) are common and abundant in Level 2 and decline markedly and almost disappear from levels 5 and 6.
- Intermediate taxa (Attribute IV) increase to high relative richness and relative abundance at BCG Level 4, but decline in Levels 5 and 6.
- Tolerant taxa (attribute V) increase in abundance and dominance at BCG levels 4 to 6, although they are represented at all levels.

Cold and coolwater invertebrates (Figure 12) - Least-disturbed coldwater streams have somewhat lower taxa richness than warmwater streams, and total taxa richness increases somewhat at BCG Level 3. Other attributes and metrics are similar between cold and warm water.

Warmwater fish (Figures 13-15):

- Taxa richness declines from BCG Level 1 to Level 6. All Level 1 sites were large waterbodies (rivers), and so may be more influenced by size than by condition

- Attribute I taxa were characteristic of BCG Level 1 (but all Level 1 sites were large rivers), and are generally absent in levels 3-6
- All sensitive taxa (attributes I, II, and III combined) are common and abundant in Levels 1 and 2 and decline markedly and almost disappear from levels 5 and 6.
- Intermediate taxa (Attribute IV) are nearly constant throughout the gradient, but decline in Level 6.
- Highly Tolerant taxa (attribute V-a) increase in abundance, dominance and variability at BCG levels 4 to 6, although they are represented at all levels.

High variability of the fish attribute metrics in Figures 13-15 is partly the result of a mix of streams from headwaters to large rivers being represented in the figures. This variability was reduced somewhat when considering single stream types.

Cold and cool water fish (Figure 16) – taxa richness of high-quality coldwater streams is low, consisting typically of brook trout and at most one or two other species. With increasing stress, other species (some warmwater) enter the community. The number of fish species increases from coldwater to coolwater to warmwater streams. In cold- and coolwater streams, taxa richness increases from BCG levels 2 to 3, but then declines in BCG level 5. Sensitivity and tolerance attributes and metrics of cold and cool streams behave similarly to warmwater streams.

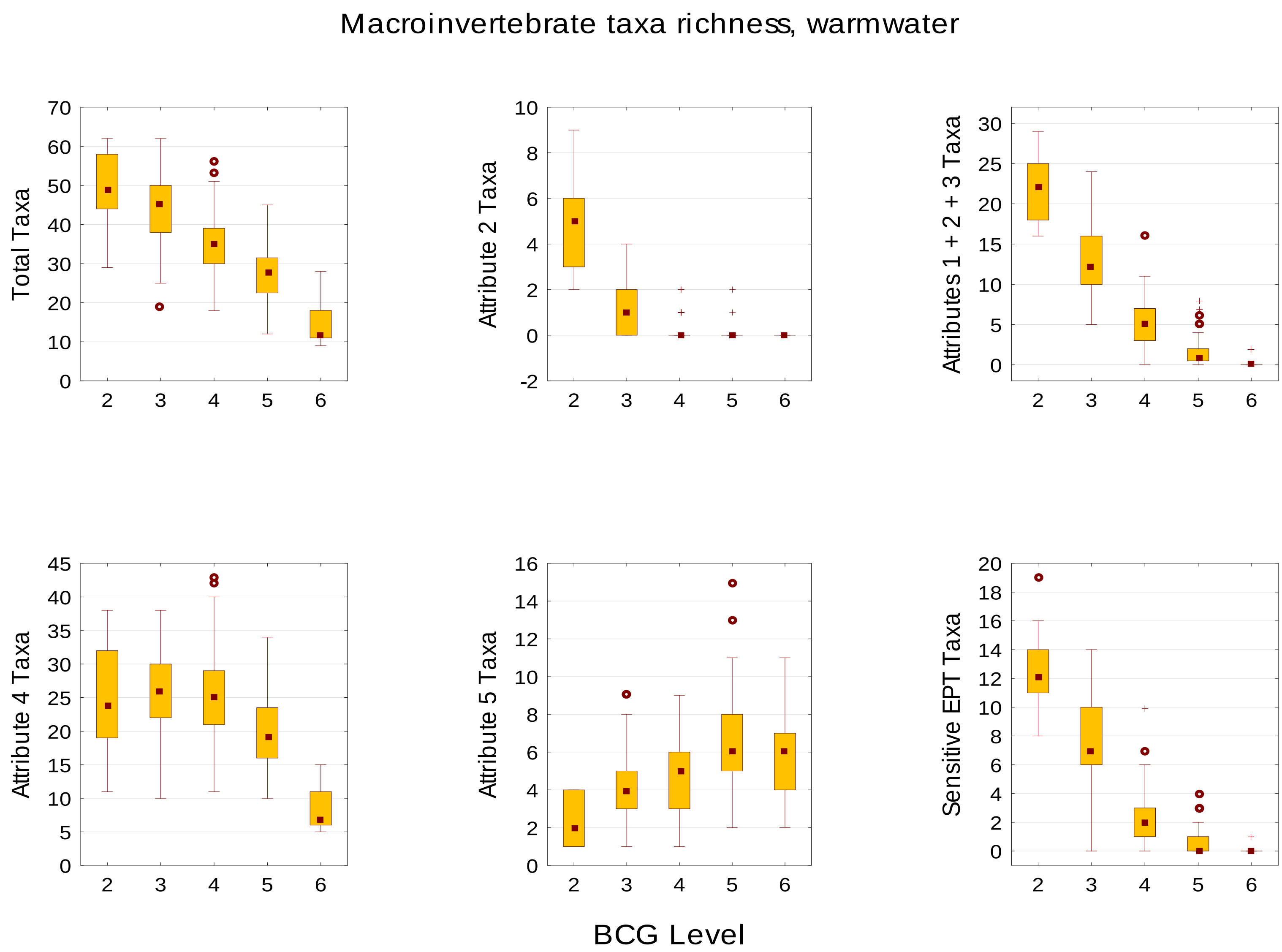

Figure 9. Benthic macroinvertebrate attribute taxa richness metrics, by BCG level (all rated warmwater sites).

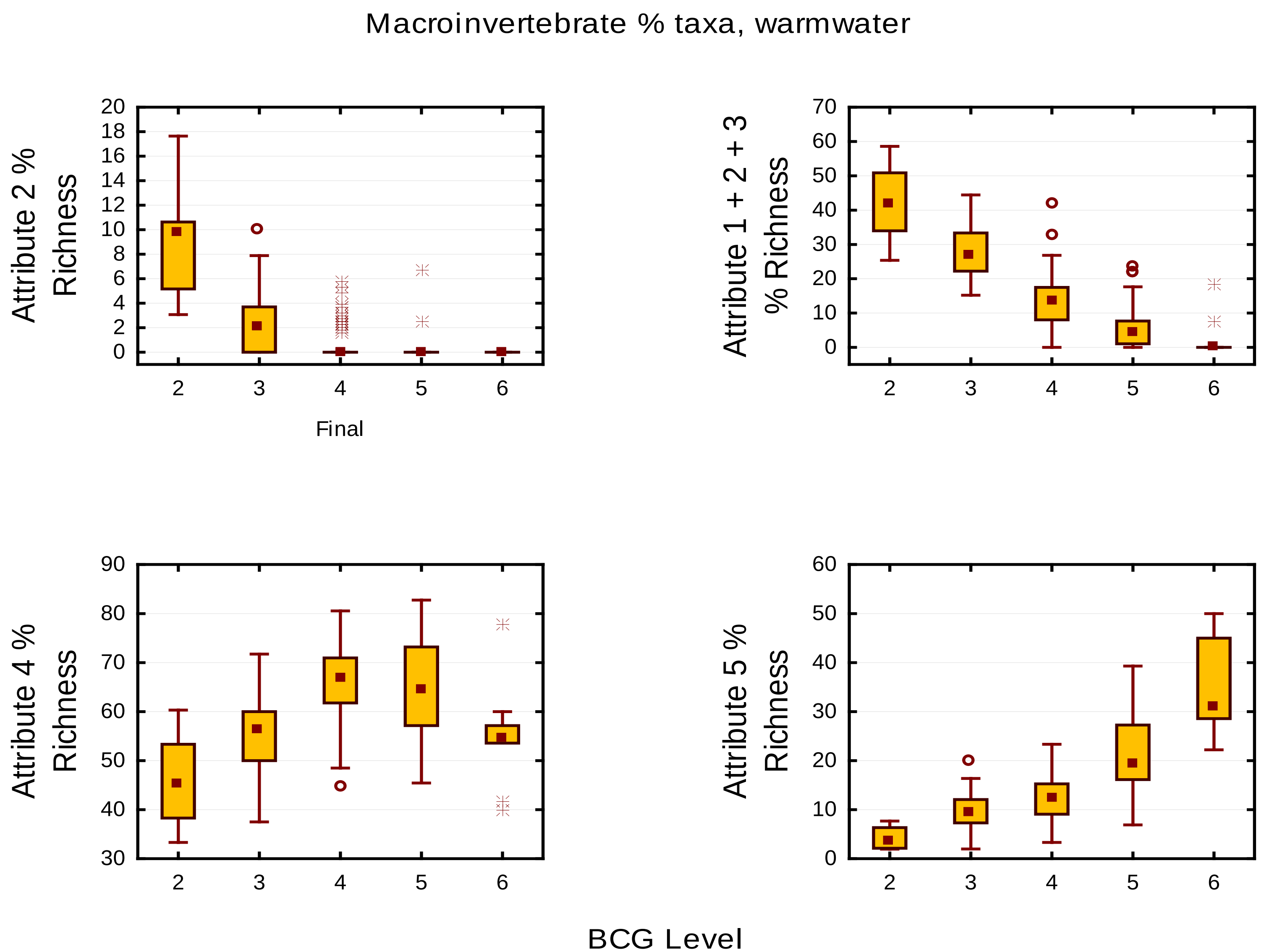

Figure 10. Benthic macroinvertebrate attribute relative richness metrics, by BCG level (all rated warmwater sites).

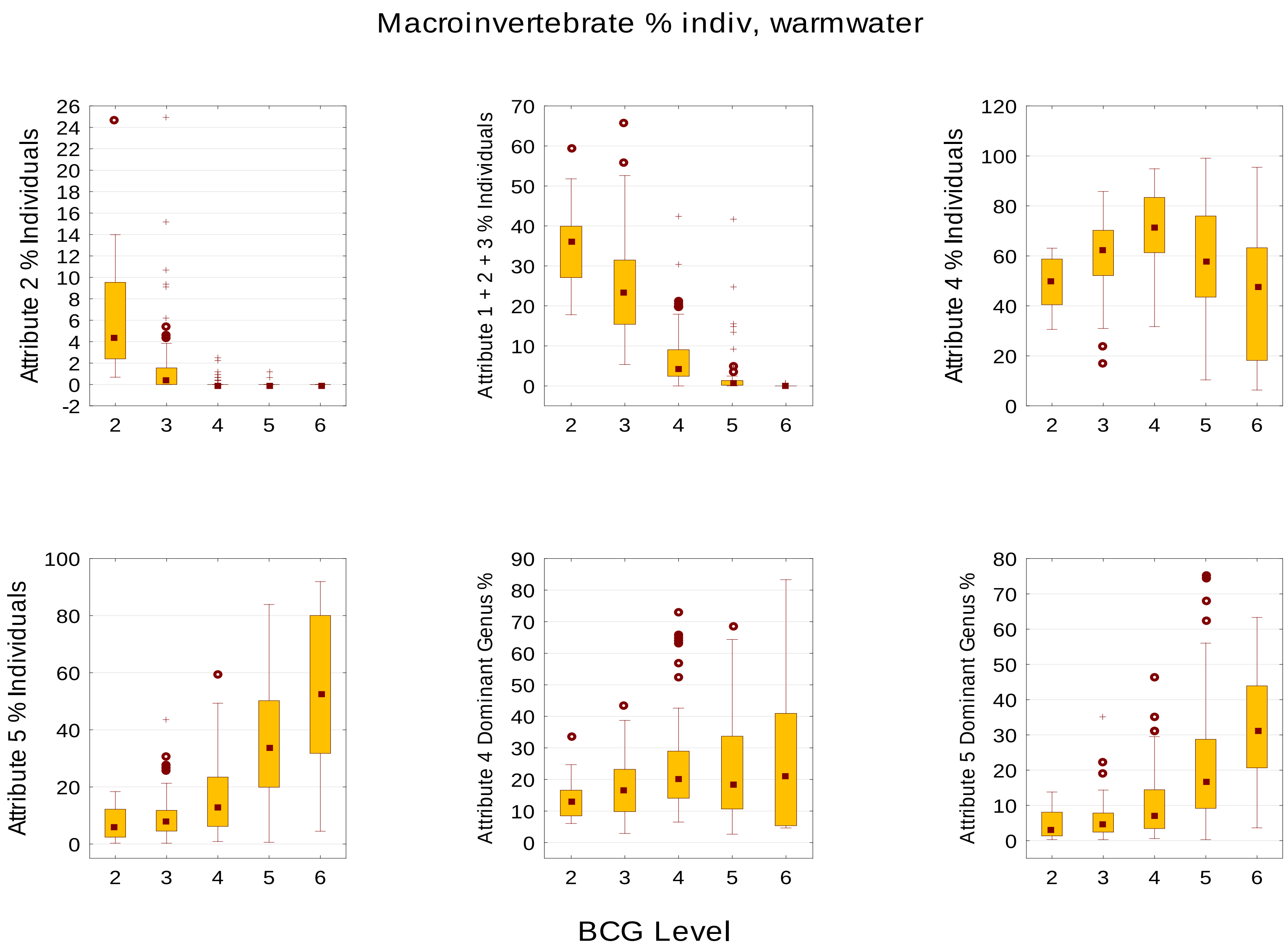

Figure 11. Benthic macroinvertebrate attribute proportional abundance and dominance metrics, by BCG level (all rated warmwater sites).

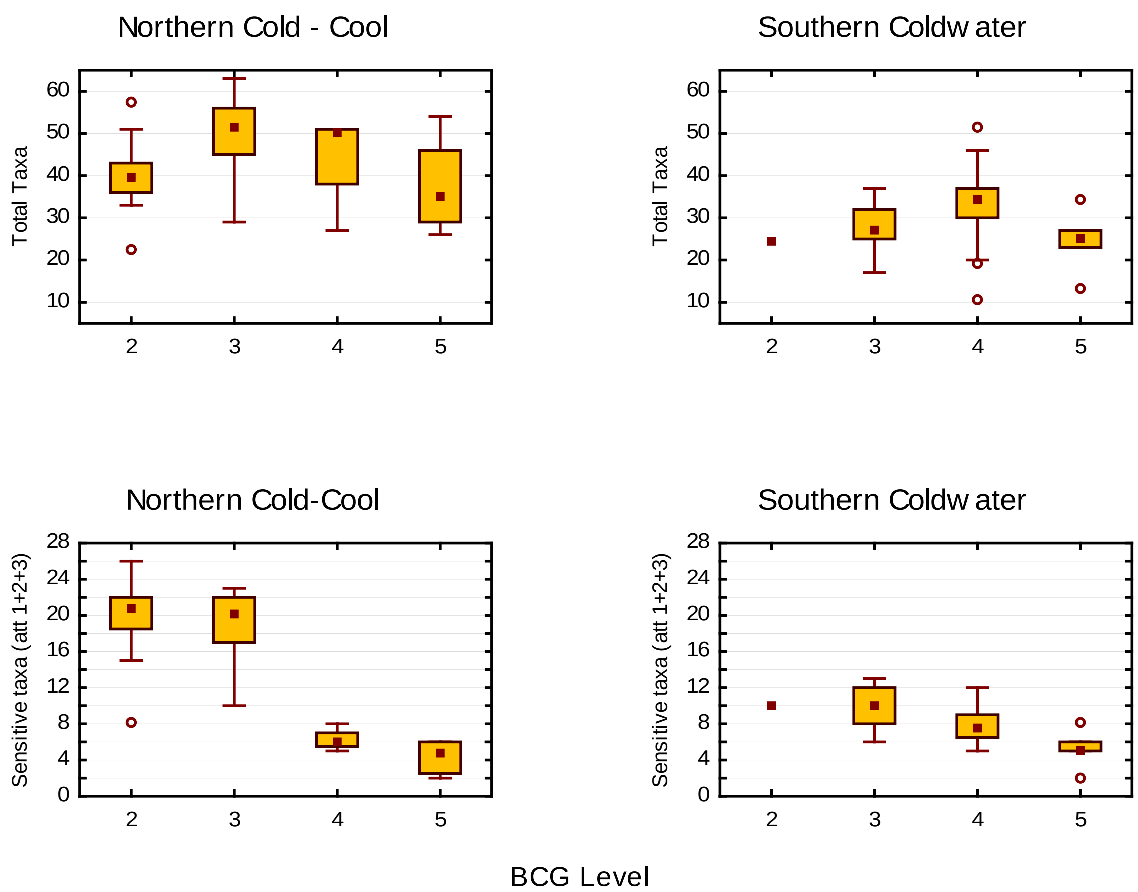

Figure 12. Selected cold and coolwater benthic macroinvertebrate metrics, by BCG level (all rated cold and coolwater sites in Minnesota).

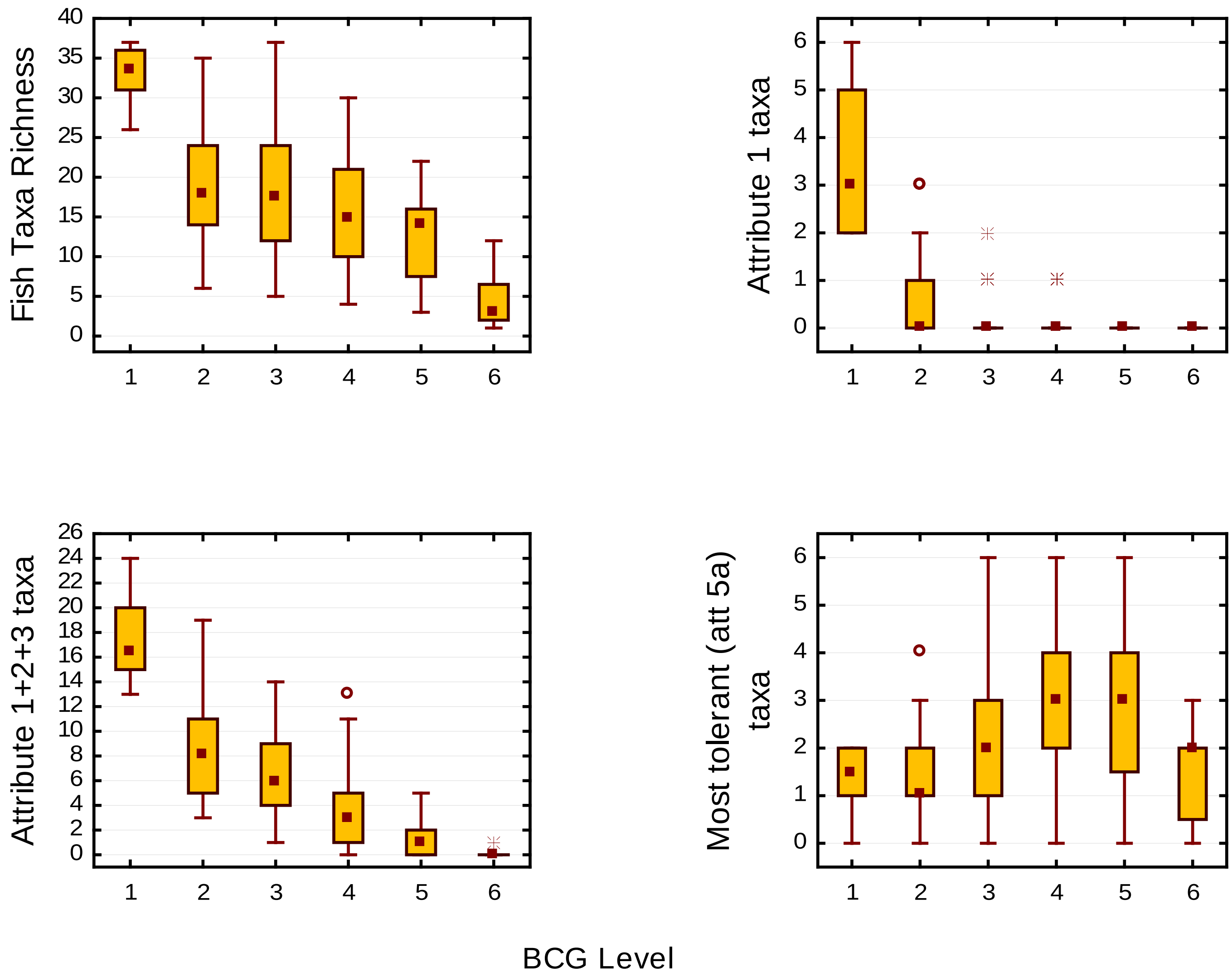

Figure 13. Fish attribute taxa richness metrics, by BCG level (all rated warmwater sites).

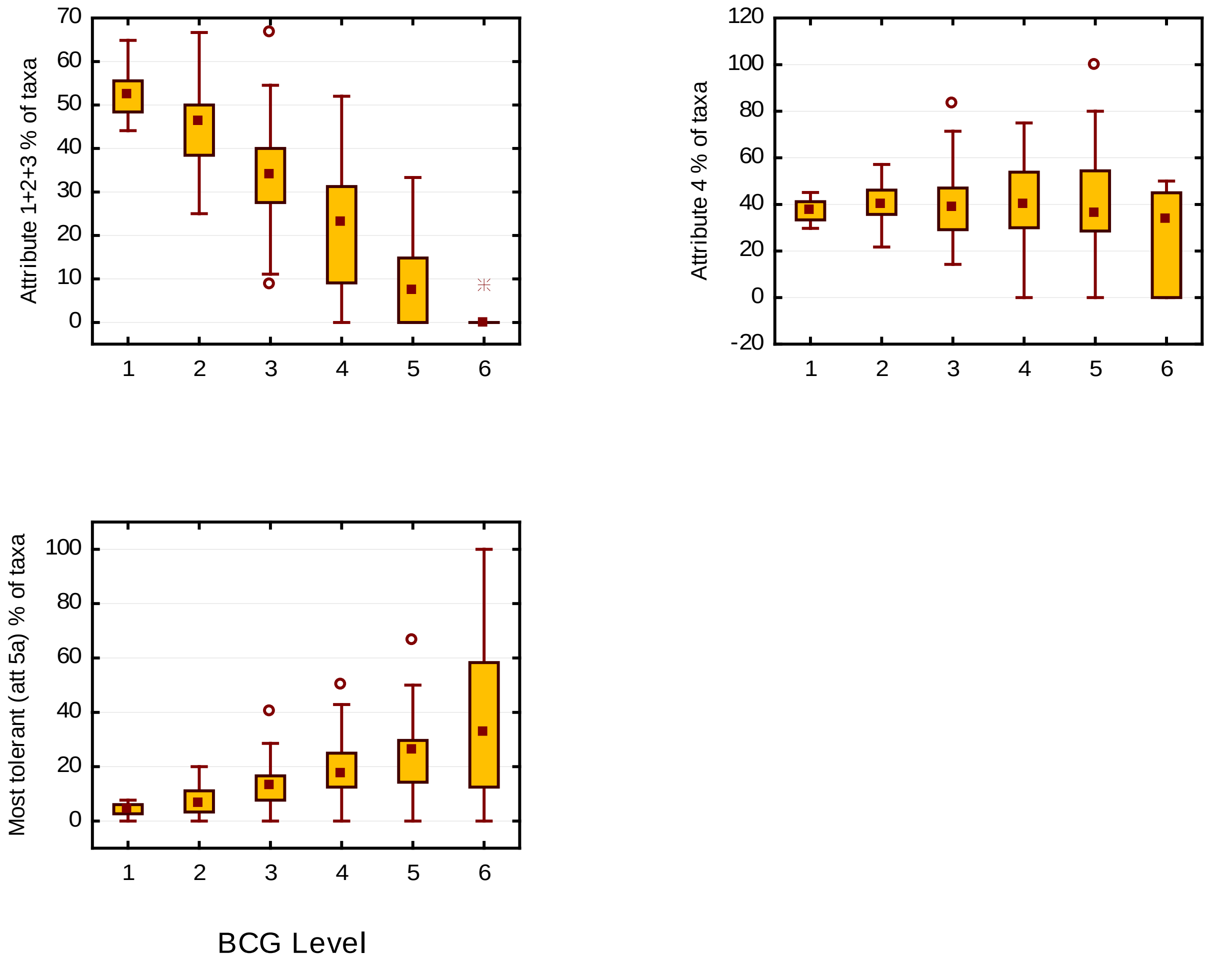

Figure 14. Fish attribute relative richness metrics, by BCG level (all rated warmwater sites).

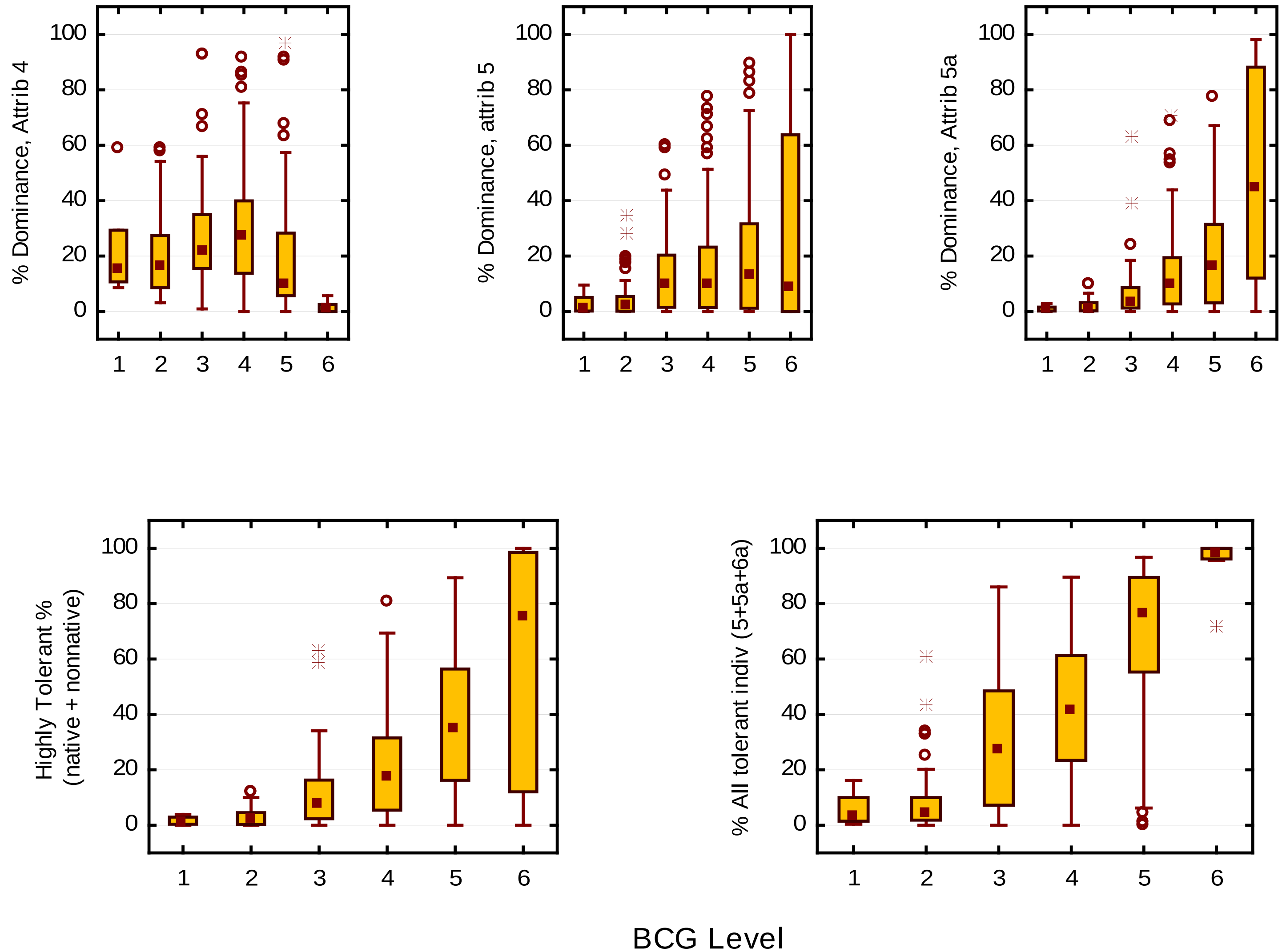

Figure 15. Fish attribute proportional abundance and dominance metrics, by BCG level (all rated warmwater sites).

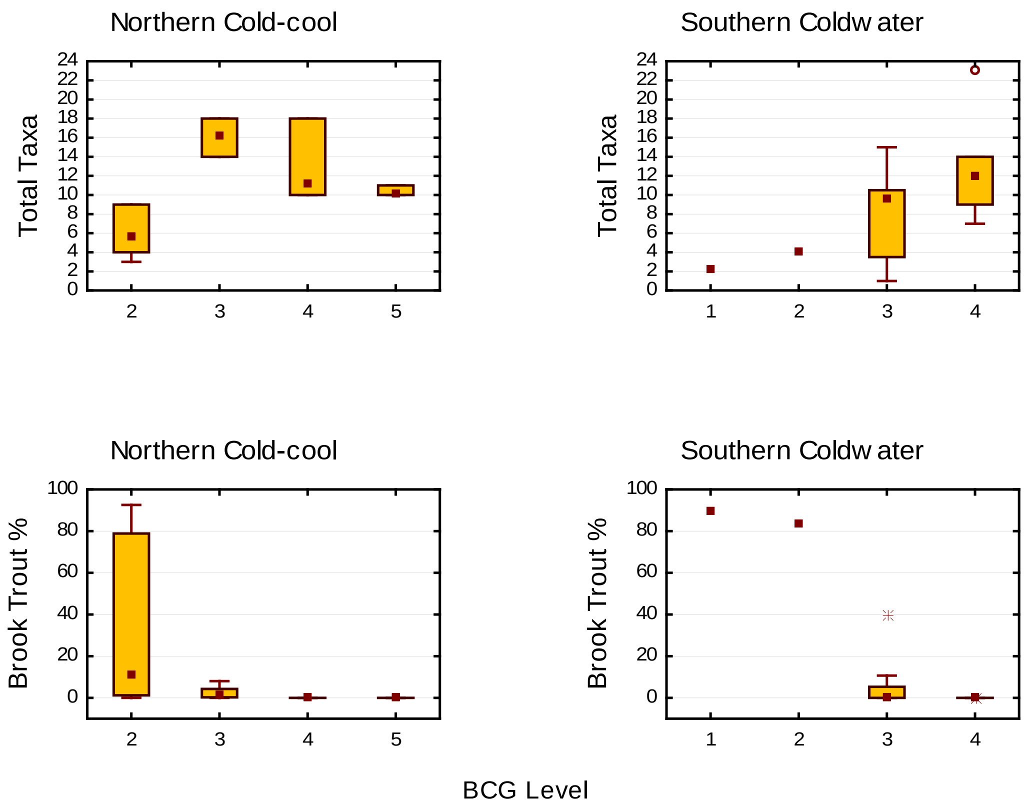

Figure 16. Selected cold and coolwater fish metrics, by BCG level (all rated cold and coolwater sites in Minnesota).

4.0 MINNESOTA BCG

4.1 BCG Rule Development

Panelists followed the descriptions of the BCG levels given in Chapter 1, and gave their reasoning during the deliberations for assigning sites to given levels. These resulted in statements such as, “This sample represents Level 4 because sensitive taxa are severely reduced but still present;” or “attribute IV and V individuals greatly outnumber sensitive individuals.” When panelists agreed on such statements they were used as preliminary rules. Initial quantitative boundaries on the rules were taken from the distributions of attribute metrics in the assigned BCG levels (Figures 9-16; Appendix C). In subsequent sessions the rules were refined by examining more samples and by re-examining samples where the panel and the candidate rules had not resulted in the same outcome. Final rules for all 18 assessed stream classes are shown in tables 5-13. The cold- and coolwater rules have been modified from Gerritsen and Stamp (2012).

In the decision model, rules work as a logical cascade from BCG level 1 to level 6. A sample is first tested against the level 1 rules; if a single rule fails, then the level fails, and the assessment moves down to level 2, and so on (Figure 17). All required rules must be true for a site to be assigned to a level. Level 6 is not listed, because failure at level 5 results in a level 6 assessment.

As described in Section 2.1, membership functions had to be defined for metrics used in the quantitative models. Membership functions are defined in the rules tables as piecewise linear functions (line segments; Figure 3), and they tend to be inequalities (“number of taxa greater than 20”). Rules in Tables 5-13 are expressed as an inequality and a range, e.g., “> 15 - 25,” where the range describes the linear segment as it increases from 0 to 1 for “>” and decreases from 1 to 0 for “<”. So, for a rule expressed as “> 15 - 25 %”, the given membership is 0 at a metric value <=15%; rises linearly to 1 at a metric value of 25%; and remains 1 for values > 25%.

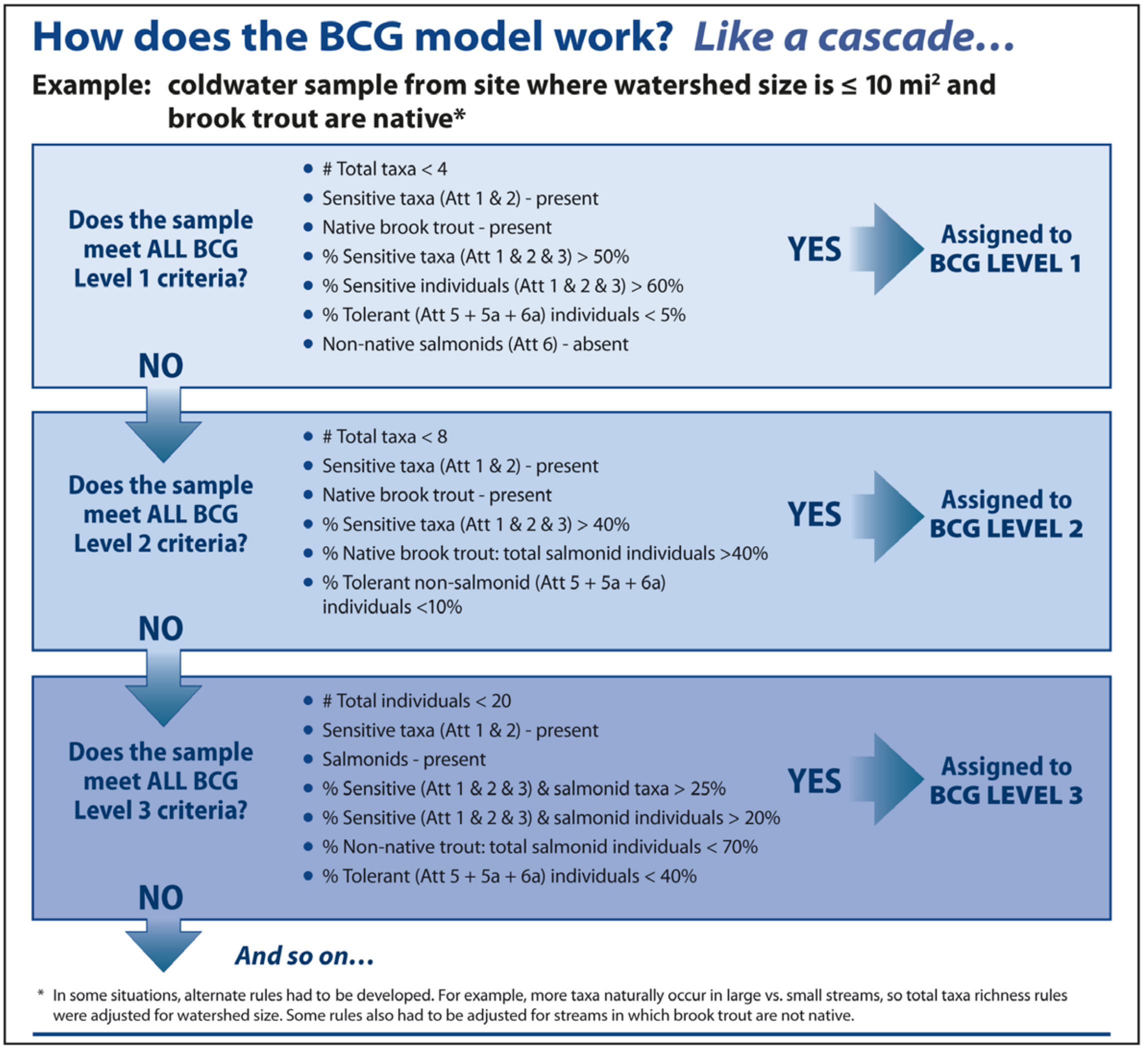

Figure 17. Flow chart depicting how rules work as a logical cascade in the BCG model. Illustration taken from Gerritsen and Stamp (2011 draft); is not identical to coldwater model in Table 8.

Some rule sets include alternatives, that is, there may be two or three alternative rules for a certain BCG level (e.g., Table 5). In this case, at least one of the alternatives must be true for the site to be assigned to that level. Alternatives usually reflected a trade-off specified by the panel: for example, a high number of total taxa could offset a low proportion of sensitive taxa, and vice-versa, to be considered (say) BCG Level 3.

In general, panelists preferred to use taxa richness within the sensitive attributes as the most important criteria for determining site BCG level assignments. Thus, the number of sensitive taxa was most often used to distinguish between BCG level 2 and level 3 sites. BCG level 2 should have several highly sensitive taxa (attribute II), but their richness may be reduced in level 3. All of the Level 1 fish samples had 2 or more Attribute I taxa (rare or endemic taxa).

The higher BCG levels all required some minimum quantities or relative richness of sensitive taxa (attributes I, II and III). These included number of taxa, percent of taxa, or percent of individuals. Additionally, for a site to be considered in Level 1 to Level 3, participants often also placed upper limits on the abundance and richness of tolerant taxa, especially abundance and dominance of attribute V. In summary, to be rated in Levels 1 to 3, sites require a minimum richness and sometimes minimum relative abundance (“floor”) of sensitive taxa (attributes I to III), and a maximum abundance and sometimes maximum richness (“ceiling”) of tolerant taxa (attribute V).

There was consistency of attribute metric values, and hence of the rules among the macroinvertebrate stream classes (Tables 10-13). The exceptions to the overall consistency were the glide-pool habitat at BCG level 2, where a greater abundance of tolerant taxa were allowed, and the coldwater streams, which have generally lower expectations of total richness in BCG levels 2 and 3 (Table 13).

Attribute values and the rules were less consistent among the fish stream types. This was in part because overall fish taxa richness is lower than invertebrate richness, and also because richness is strongly dependent on stream size. Headwater streams and wetland-lacustrine streams were relatively depauperate, which results in poorer precision and discriminatory ability of any index or assessment method that uses the fish assemblage data in these habitats.

Table 5. Decision rules for fish assemblages in rivers. Rules show the ranges of fuzzy membership functions (see Fig. 9). N shows the number of sites at the indicated BCG level and stream class in the calibration data set.

Metric	Prairie Rivers (1)	Northern Forest Rivers (4)		Wetland-Lacustrine (7)	
BCG Level 1	N=2	N=3		N=0[1]	
Total taxa	> 25 - 35	> 16 - 24		> 25 - 35	
Endemic taxa (Att 1)	Present	Present		Present	
Att 1+2 taxa	> 2 - 5	> 1 - 2		> 2 - 5	
Att 1+2+3 % taxa	> 45 - 55%	> 35 - 45%		> 45 - 55%	
Att 1+2+3 % ind	> 25 - 35%	> 45 - 55%		> 25 - 35%	
Att 5a or 6a Dominance		< 7 - 13%			
Tolerant % ind (5 + 5a + 6a)	< 3 - 7%			< 3 - 7%	
Highly tol % ind (5a + 6a)		< 7 - 13%			
BCG Level 2	N=6	N=15		N=7	
				Alt 1	**Alt 2**
Total taxa	> 16 - 24	> 6 - 10		> 6 - 10	> 11 - 16
Att 1+2 taxa	Present			Present	n/a
Att 1+2+3 % taxa	> 35 - 45%	> 25 - 35%		> 25 - 35%	= alt 1[2]
Att 1+2+3 % Ind	> 15 - 25%	> 25 - 35%		> 30 - 40%	= alt 1[2]
Att 5a or 6a Dominance		< 7 - 13%			
Highly tol % ind (5a + 6a)	< 7 - 13%	< 7 - 13%		< 7 - 13%	= alt 1[2]
BCG Level 3	N=25	N=11		N=7	
				Alt 1	**Alt 2**
Total taxa	> 11 - 16	> 6 - 10		> 1 - 5	> 6 - 10
Att 1+2+3 % taxa	> 15 - 25%	> 15 - 25%		> 10 - 20%	> 20 - 30%
Att 1+2+3 % Ind	> 7 - 13%	> 7 - 13%		> 10 - 20%	> 20 - 30%
Tol % ind (5 + 5a + 6a)		< 25 - 35%			
Att 5a or 6a Dominance	< 7 - 13%	< 10 - 20%			
Highly tol % ind (5a + 6a)	<25 - 35%			< 7 - 13%	<35 - 45%
BCG Level 4	N=31	N=16		N=11	
		Alt 1	**Alt 2**	**Alt 1**	**Alt 2**
Total taxa	> 11 - 16	> 6 - 10	= alt 1[2]	> 1 - 5	> 6 - 10
Att 1+2+3 % taxa	10 - 20%	> 15 - 25%	> 7 - 13%	present	> 7 - 13%
Att 1+ 2+3 % Ind	0 - 1%	> 3 - 7%	present	n/a	> 7 - 13%
1+2+3+4 % Ind				> 45 - 55%	n/a
Att 5a or 6a Dominance	<35 - 45%	< 25 - 35%	= alt 1[2]	< 35 - 45%	<45 - 55%
Tol % ind (5 + 5a + 6a)		n/a	< 30 - 40%		
Highly Tol % ind (5a + 6a)	<45 - 55%	< 35 - 45%	= alt 1[2]		
BCG Level 5	N=12	N=2		N=6	
Total taxa	> 11 - 16	6 - 10		>0 - 4	
Att 1+2+3+4 % Taxa				present	
Att 5a or 6a Dominance	<65 - 75%	<35 - 45%		<55 - 65%	
Highly tol % ind (5a + 6a)		<55 - 65%			
BCG Level 6 (no rules)	N=1	N=0		N=2	

[1] **BCG Level 1 for Wetland-lacustrine (shaded) set to same criteria as Prairie Rivers.**

[2] "= alt 1" the rule is the same as given under Alt 1 for this metric

Table 6. Decision rules for fish assemblages in wadeable streams, as in Table 5.

Metric	Southern Wadeable Streams (2)			Northern Wadeable Streams (5)	
BCG Level 1	N=0[1]			N=0[1]	
total taxa	> 25 - 35			> 25 - 35	
1 Endemic taxa	present			present	
Att 1+2 taxa	>2 - 5			>2 - 5	
att 1+2+3 % taxa	> 45 - 55%			> 45 - 55%	
att 1+2+3 % Ind	> 25 - 35%			> 25 - 35%	
Tol % ind (5 + 5a + 6a)	< 3 - 7%			< 3 - 7%	
BCG Level 2	N=1			N=8	
total taxa	> 16 - 24			>11 - 16	
att 1+2+3 total taxa	> 6 - 10				
att 1+2+3 % taxa	> 35 - 45%			> 25 - 35%	
att 1+2+3 % Ind	> 7 - 13%			> 7 - 13%	
att 5a or 6a dom				< 7 - 13%	
Tol % ind (5 + 5a + 6a)				< 30 - 40%	
Highly tol % ind (5a + 6a)	< 15 - 25%				
BCG Level 3	N=4			N=10	
total taxa	>11 - 16			> 11 - 16	
att 1+2+3 % taxa	> 7 - 13%			> 20 - 30%	
att 1+2+3 % Ind	> 3 - 7%			> 3 - 7%	
att 5a or 6a dom	< 15 - 25%			< 7 - 13%	
Highly tol % ind (5a + 6a)	< 35 - 45%			< 15 - 25%	
BCG Level 4	N=10			N=15	
	Alt 1	**Alt 2**		**Alt 1**	**Alt 2**
total taxa	> 6 - 10	> 16 - 24		> 6 - 10	= alt 1[2]
att 1+2+3 % taxa	0 - 1%	n/a		> 3 - 7%	n/a
att 1+ 2+3 % Ind	0 - 1%	n/a		present	n/a
1+2+3+4 % Ind				n/a	> 65 - 75%
att 1+2+3+4 % taxa				n/a	> 45 - 55%
att 5a or 6a dom	< 45 - 55%	= alt 1[2]		< 25 - 35%	< 15 - 25%
Tol % ind (5 + 5a + 6a)	<65 - 75%	= alt 1[2]			
Highly tol % ind (5a + 6a)	<55 - 65%	= alt 1[2]		<55 - 65%	n/a
BCG Level 5	N=18			N=4	
	Alt 1	**Alt 2**	**Alt 3**		
total taxa	> 3 - 7	> 11 - 16	> 16 - 24	>1 - 5	
att 1+2+3 % Taxa	n/a	present	n/a		
att 1+2+3+4 % Taxa	> 7 - 13%	n/a	> 15 - 25%	> 10 - 20%	
att 5a or 6a dom	< 45 - 55%	n/a	n/a	< 65 - 75%	
Highly tol % ind (5a + 6a)	< 65 - 75%	n/a	n/a		
BCG Level 6 (no rules)	N=2			N=0	

[1]**BCG Level 1 (shaded) set to same criteria as Prairie Rivers, Table 4-1.**

[2] "= alt 1" the rule is the same as given under Alt 1 for this metric

Table 7. Decision rules for fish assemblages in headwater streams, as in Table 5.

Metric	**Southern Headwaters (3)**		**Northern Headwaters (6)**		
BCG Level 1	N=0[1]		N=0[1]		
total taxa	> 25 - 35		> 25 - 35		
1 Endemic taxa	present		present		
Att 1+2 taxa	>2 - 5		>2 - 5		
att 1+2+3 % taxa	> 45 - 55%		> 45 - 55%		
att 1+2+3 % Ind	> 25 - 35%		> 25 - 35%		
Tol % ind (5 + 5a + 6a)	< 3 - 7%		< 3 - 7%		
BCG Level 2	N=0		N=4		
total taxa	> 6 - 10		> 6 - 10		
att 1+2+3 total taxa	> 0 - 4		> 1 - 4		
att 1+2+3 % taxa	>15 - 25%		>15 - 25%		
att 1+2+3 % Ind	> 15 - 25%		> 15 - 25%		
att 5a or 6a dom	< 3 - 7%		< 3 - 7%		
Highly tol % ind (5a + 6a)	< 7 - 13%		< 7 - 13%		
BCG Level 3	N=3		N=9		
total taxa	> 5 - 9		> 3 - 7		
att 1+2+3 % taxa	present		> 10 - 20%		
att 1+2+3 % Ind			> 7 - 13%		
att 1+2+3+4 % taxa	15 - 25%				
att 5a or 6a dom	< 3 - 7%		< 25 - 35%		
Highly tol % ind (5a + 6a)	< 7 - 13%		< 25 - 35%		
BCG Level 4	N=22		N=10		
	Alt 1	**Alt 2**	**Alt 1**	**Alt 2**	**Alt 3**
total taxa	> 4 - 8	= alt 1[2]	> 6 - 10	> 2 - 5	present
att 1+2+3 % taxa	n/a	present	> 7 - 13%	= alt 1[2]	= alt 1[2]
att 1+ 2+3 % Ind			> 3 - 7%	= alt 1[2]	= alt 1[2]
att 1+2+3+4 % taxa	> 7 - 13%	= alt 1[2]			
att 5a or 6a dom	< 45 - 55%	n/a	< 35 - 45%	<25 - 35%	absent
Highly tol % ind (5a + 6a)					
BCG Level 5	N=4		N=8		
total taxa	> 1 - 5		> 0 - 4		
att 1+2+3+4 % Taxa			> 7 - 13%		
att 5a or 6a dom	< 65 - 75%				
BCG Level 6 (no rules)	N=3		N=0		

[1]BCG Level 1 for Wetland-lacustrine (shaded) set to same criteria as Prairie Rivers.
[2] "= alt 1" the rule is the same as given under Alt 1 for this metric

Table 8. Decision rules for fish assemblages in southern coldwater streams (Driftless area in MN). Modified from Gerritsen and Stamp (2013). Numbers (N) include sites in Wisconsin and Michigan.

Metric	**Southern Coldwater (10)**			
BCG Level 1	N=4			
	Brook Trout native		**Brook trout not native**	
Total taxa	< 2 - 5		= alt 1[1]	
Brook trout	present		absent	
Att 1+2 taxa	0 - 1		= alt 1[1]	
Att 1+2+3 % taxa	> 45 - 55%		= alt 1[1]	
Att 1+2+3 % Ind	> 55 - 65%		= alt 1[1]	
Other Salmonidae (nonnative)	absent		= alt 1[1]	
Tolerant% ind (5 + 5a + 6a)	< 3 - 7%		= alt 1[1]	
BCG Level 2	N=9			
	Brook Trout native		**Brook trout not native**	
	Alt 1	**Alt 2**	**Alt 1**	**Alt 2**
Total taxa (by area)	if area < 10, (< 6-10), else (> 2-5 AND < 11-16)			
Brook trout % ind	present	= alt 1[1]	n/a	n/a
Att 1+2+3 % taxa	> 35 - 45%	> 15 - 25%	n/a	> 15 - 25%
Att 1+2+3+6 % Ind	n/a	n/a	> 65 - 75%	n/a
BT % of total Salmonidae	> 35 - 45%	= alt 1[1]	n/a	n/a
Tolerant% ind (5 + 5a + 6a)	< 7 - 13%	< 0 - 1%	n/a	< 7 - 13%
BCG Level 3	N=17; BT status not relevant for Levels 3 - 6			
	Alt 1		**Alt 2**	
Number individuals (by area)				
Att 1+2 taxa	n/a		0 - 1	
sensitive + Salmonidae % taxa	20 - 30%		= alt 1[1]	
sensitive + Salmonidae % Ind	15 - 25%		= alt 1[1]	
BT + Att 6 % ind (all trout)	0 - 1%		= alt 1[1]	
Att 4-5 dom	< 45 - 55%		= alt 1[1]	
Tolerant% ind (5 + 5a + 6a)	< 7 - 13%		< 35 - 45%	
BCG Level 4	N=9			
Att 1+2+3+6 % taxa	3 - 7%			
Att 1+2+3+6 % Ind	3 - 7%			
% Taxa (5 + 5a + 6a)	< 40 - 50%			
Highly Tolerant % ind (5a + 6a)	< 7 - 13%			
BCG Level 5	N=8			
Total taxa	> 1 - 4			
Att 1+2+3+4 % Taxa	> 7 - 13%			
BCG Level 6 (no rules)	N=0			

[1] **"= alt 1": the rule is the same as given under Alt 1 for this metric**

Table 9. Decision rules for fish assemblages in northern cold-cool water streams. Modified from Gerritsen and Stamp (2011). Numbers (N) include sites in Wisconsin and Michigan.

Metric	Northern Cold-cool (11)	
	N=0	
BCG Level 1	**Brook Trout native**	**Brook trout not native**
Total taxa	> 2 - 5 and < 11 - 16	= alt 1[1]
Brook trout	present	absent
Att 1+2 taxa	0 - 1	= alt 1[1]
Att 1+2+3 % taxa	> 35 - 45%	= alt 1[1]
Att 1+2+3 % Ind	> 35 - 45%	= alt 1[1]
Other Salmonidae (nonnative)	absent	= alt 1[1]
Tolerant % ind (5 + 5a + 6a)	< 3 - 7%	= alt 1[1]
BCG Level 2	N=14	
total taxa (by area)	< 16 - 24	= alt 1[1]
Brook trout % ind	present	n/a
Att 1+2 taxa	0 - 1	n/a
Att 1+2+3 % taxa	> 25 - 35%	= alt 1[1]
Att 1+2+3 % Ind	> 17 - 27%	= alt 1[1]
BT % of total Salmonidae	> 35 - 45%	n/a
Tolerant % ind (5 + 5a + 6a)	<15 - 25%	= alt 1[1]
BCG Level 3	N=13; BT status not relevant for Levels 3 - 6	
	Alt 1	**Alt 2**
Number individuals (by area)		
Total taxa	<16 - 24	= alt 1[1]
Sensitive + Salmonidae % taxa	Sensitive + Salmonidae % taxa > tolerant % taxa (Att 5, 5a, 6a)	n/a
Sensitive + Salmonidae % Ind	n/a	Sensitive + Salmonidae % ind > tolerant % ind (Att 5, 5a, 6a)
Att 4-5 dom	IF area > 5, THEN < 60 - 70	= alt 1[1]
Tolerant% ind (5 + 5a + 6a)		
Highly tolerant % ind (5a + 6a)	<3 - 7%	= alt 1[1]
BCG Level 4	N=9	
Att 1+2+3+6 % taxa	> 3 - 7%	
Highly Tolerant % ind (5a + 6a)	< 15 - 25%	
BCG Level 5	N=6	
Total taxa	> 1 - 4	
Att 1+2+3+4 % Taxa	> 7 - 13%	
BCG Level 6 (no rules)	N=0	

[2] **"= alt 1" the rule is the same as given under Alt 1 for this metric**

Table 10. Decision rules for macroinvertebrate assemblages in rivers, as in Table 5.

Metric	Prairie Rivers (2)	Northern Forest Rivers (1)	
BCG Level 2	N=0	N=7	
Total taxa	> 35 - 45	> 35 - 45	
Att 1+2 taxa	> 2 - 5	> 1 - 4	
Att 1+2+3 % taxa	> 20 - 30%	> 20 - 30%	
Att 1+2+3 % Ind	> 10 - 20%	> 10 - 20%	
Att 5 % Ind	< 7 - 13%	< 7 - 13%	
Sensitive EPT taxa	> 6 - 10	> 6 - 10	
BCG Level 3	N=6	N=15	
		Alt 1	**Alt 2**
Total taxa	> 25 - 35	> 20 - 30	> 40 - 50
Att 1+2+3 % taxa	> 10 - 20%	> 15 - 25%	> 7 - 13%
Att 1+2+3 % Ind	> 3 - 7%	> 7 - 13%	> 3 - 7%
Att 5 % Ind	< 15 - 25%	< 35 - 45%	= alt 1[1]
Att 5 Dom	< 10 - 20%	< 25 - 35%	= alt 1[1]
Sensitive EPT taxa	> 2 - 5	> 2 - 5	= alt 1[1]
BCG Level 4	N=19	N=6	
Total taxa	> 16 - 24	> 16 - 24	
Att 1+2+3 % taxa	> 3 - 7%	> 7 - 13%	
Att 1+2+3 % Ind	present	> 3 - 7%	
Att 5 % Ind	< 45 - 55%	< 45 - 55%	
Att 5 Dom	< 35 - 45%	< 35 - 45%	
Sensitive EPT taxa	present	present	
BCG Level 5	N=4	N=0	
Total taxa	> 16 - 24	> 16 - 24	
Att 5 % taxa	< 35 - 45%	< 35 - 45%	
Att 5 Dom	< 65 - 75	< 65 - 75	
BCG Level 6 (no rules)	N=0	N=0	

[2] "= alt 1" the rule is the same as given under Alt 1 for this metric

Table 11. Decision rules for macroinvertebrate assemblages in riffle-run habitat, as in Table 5.

Metric	**5 Southern riffle-run**		**3 Northern forest riffle-run**	
BCG Level 2	N=0		N=2	
Total taxa	> 35 - 45		> 35 - 45	
Att 1+2 taxa	> 2 - 5		> 2 - 5	
Att 1+2+3 % taxa	> 45 - 55%		> 45 - 55%	
Att 1+2+3 % Ind	> 25 - 35%		> 25 - 35%	
Att 5 % Ind	< 3 - 7%		< 7 - 13%	
Sensitive EPT taxa	> 11-16		> 9 - 14	
BCG Level 3	N=8		N=17	
	Alt 1	**Alt 2**	**Alt 1**	**Alt 2**
Total taxa	> 25 - 35	> 40 - 50	> 25 - 35	> 40 - 50
Att 1+2+3 % taxa	> 15 - 25%	> 7 - 13%	> 15 - 25%	> 10 - 20%
Att 1+2+3 % Ind	> 10 - 20%	> 3 - 7%	> 7 - 13%	> 3 - 7%
Att 4 Dom			< 20 - 30%	= alt 1[1]
Att 5 % Ind	< 15 - 25%	= alt 1[1]		
Att 5 Dom	< 7 - 13%	= alt 1[1]	< 30 - 40%	= alt 1[1]
Sensitive EPT taxa	> 2 - 5	= alt 1[1]	> 2 - 5	= alt 1[1]
BCG Level 4	N=19		N=9	
	Alt 1	**Alt 2**		
Total taxa	> 16 - 24	> 25 - 35	> 16 - 24	
Att 1+2+3 % taxa	> 3 - 7%	present	> 7 - 13%	
Att 1+2+3 % Ind	> 3 - 7%	present	present	
Att 5 % Ind	< 30 - 40%	< 35 - 45%	< 30 - 40%	
Att 5 Dom	< 15 - 25%	= alt 1[1]	< 20 - 30%	
Sensitive EPT	present	= alt 1[1]	present	
BCG Level 5	N=20		N=2	
	Alt 1	**Alt 2**	**Alt 1**	**Alt 2**
Total taxa	> 11 - 16	> 16 - 24	> 11 - 16	> 16 - 24
Att 2+3+4 % taxa	n/a	> 45 - 55%		
Att 5 % taxa	< 35 - 45%	n/a	< 35 - 45%	< 45 - 55%
Att 5 Dom	< 55 - 65%	n/a	< 55 - 65%	= alt 1[1]
BCG Level 6 (no rules)	N=0		N=0	

[2] "= alt 1" the rule is the same as given under Alt 1 for this metric

Table 12. Decision rules for macroinvertebrate assemblages in glide-pool habitat, as in Table 5.

Metric	**7 Prairie glide-pool**		**6 Southern forest glide-pool**		**4 Northern Forest glide-Pool**
BCG Level 2	N=0		N=0		N=5
Total taxa	> 25 - 35		> 25 - 35		> 20 - 30
Att 1+2 taxa	present		present		present
Att 1+2+3 % taxa	> 25 - 35%		> 25 - 35%		> 25 - 35%
Att 1+2+3 % Ind	> 15 - 25%		> 15 - 25%		> 15 - 25%
Att 4 Dom	< 10 - 20%		< 10 - 20%		< 10 - 20%
Att 5 % Ind	< 15 - 25%		< 15 - 25%		< 15 - 25%
Sensitive EPT taxa	> 6-10		> 6-10		> 6-10
BCG Level 3	N=3		N=5		N=13
	Alt 1	**Alt 2**	**Alt 1**	**Alt 2**	
Total taxa	> 25 - 35	> 40 - 50	> 14 - 22	> 25 - 35	> 16 - 24
Att 1+2+3 % taxa	> 10 - 20%	= alt 1[1]	> 10 - 20%	> 7 - 13%	> 10 - 20%
Att 1+2+3 % Ind	> 3 - 7%	present	> 3 - 7%	present	> 3 - 7%
Att 4 Dom			< 45 - 55%	= alt 1[1]	
Att 5 % Ind	< 30 - 40%	= alt 1[1]	< 15 - 25%	= alt 1[1]	< 25 - 35%
Att 5 Dom	< 10 - 20%	= alt 1[1]	< 10 - 20%	= alt 1[1]	< 15 - 25%
Sensitive EPT taxa	> 2 - 5	= alt 1[1]	present	= alt 1[1]	> 2 - 5
BCG Level 4	N=19		N=18		N=12
Total taxa	> 16 - 24		> 14 - 22		> 16 - 24
Att 1+2+3 % taxa	> 3 - 7%		> 0 - 4%		> 3 - 7%
Att 1+2+3 % Ind	present		> 0 - 2%		present
Att 5 % taxa			< 20 - 30%		
Att 5 % Ind	< 35 - 45%		< 30 - 40%		< 25 - 35%
Att 5 Dom	< 20 - 30%		< 15 - 25%		< 20 - 30%
BCG Level 5	N=26		N=13		N=2
Total taxa	> 12-20		> 11 - 16		> 11 - 16
Att 5 % taxa	< 50 - 60%		< 55 - 65%		< 35 - 45%
Att 5 Dom	< 45 - 55%		< 55 - 65%		< 55 - 65%
BCG Level 6 (no rules)	N=5		N=1		N=3

[2] **“= alt 1” the rule is the same as given under Alt 1 for this metric**

Table 13. Decision rules for macroinvertebrate assemblages in cold and cool waters. Modified from Gerritsen and Stamp (2013). Minnesota sites only

Metric	**9 Southern Coldwater**		**8 Northern Cold-cool**	
BCG Level 2	N=1		N=16	
Total taxa	> 11 - 16		> 16 - 24	
Att 1+2 taxa			> 2 - 5	
Att 1+2 % taxa	> 7 - 13%			
Att 1+2 % ind			> 4 - 10%	
Att 1+2+3 % taxa	> 25 - 35%		> 25 - 35%	
Att 1+2+3 % Ind	> 25 - 35%		> 25 - 35%	
Att 5 Dom	< 3 - 7%			
Sensitive EPT % Ind	> 7 - 13%		> 7 - 13%	
BCG Level 3	N=17		N=10	
	Alt 1	**Alt 2**	**Alt 1**	**Alt 2**
Total taxa	> 11 - 16	= alt 1[1]	> 16 - 24	= alt 1[1]
Att 1+2 taxa			present	n/a
Att 1+2+3 % taxa	> 15 - 25%	> 35 - 45%	> 15 - 25%	= alt 1[1]
Att 1+2+3 % Ind	> 7 - 13%	> 3 - 7%	> 7 - 13%	> 35 - 45%
Att 4 Dom	< 45 - 55%	= alt 1[1]		
Att 5 % Ind	< 15 - 25%	= alt 1[1]		
Att 5 Dom			< 7 - 13%	= alt 1[1]
Sensitive EPT % taxa	> 7 - 13%	= alt 1[1]	> 7 - 13%	= alt 1[1]
BCG Level 4	N=20		N=4	
Total taxa	> 6 - 10		> 11 - 16	
Att 1+2+3 % taxa	> 7 - 13%		> 7 - 13%	
Att 1+2+3 % Ind	> 3 - 7%		present	
Att 5 % Ind	< 35 - 45%		< 55 - 65%	
Sensitive EPT	present		present	
BCG Level 5	N=5		N=4	
Total taxa	> 6 - 10		> 11 - 16	
Att 5 % taxa	< 55 - 65%			
Att 5 Dom			< 55 - 65%	
BCG Level 5	N=0		N=0	

[2] **"= alt 1" the rule is the same as given under Alt 1 for this metric**

4.2 Model Performance

Model performance was compared to the panel assignments (i.e., the calibration data set), and is shown in Table 14. The initial effort included panel ratings of a smaller, independent data set to assess the model's post-calibration performance. However, these data were later used to adjust the model. Accordingly, model performance can only be judged based on the calibration data set.

The performance range of the fish models was 77 % to 89% correct, and the benthic models were 79% to 98% correct, in replicating the panel decisions. All of the model assignments were within one level of the majority panel opinion.

Table 14. Automated model performance at replicating panel decisions.

Fish			Benthic macroinvertebrates		
Stream Class	**N**	**% Correct**	**Stream Class**	**N**	**% Correct**
Prairie Rivers (1)	75	76%	Prairie Rivers (north and south) (2)	29	90%
Northern Forest Rivers (4)	47	87%	Northern Forest River (1)	37	76%
Wetland-lacustrine Streams (7)	32	84%	Southern Riffle-run (5)	47	89%
Southern Wadeable Streams (2)	35	74%	Northern Forest Riffle-run (3)	37	78%
Northern Wadeable Streams (5)	37	84%	Southern Hardwood Glide-pool (6)	37	86%
Southern Headwaters (3)	32	88%	Northern Forest Glide-pool (4)	35	86%
Northern Headwaters (6)	30	77%	Prairie Glide-pool (7)	52	87%
Southern Coldwater (10)*	47	89%	Southern Coldwater (9)*	43	98%
Northern Coolwater (11)*	42	81%	Northern Coolwater (8)*	34	79%
Total	377	82%	**Total**	351	86%

* Southern coldwater and northern coolwater were initially developed in Gerritsen and Stamp (2013), and modified here.

5.0 DISCUSSION AND CONCLUSIONS

The Minnesota BCG is promising as a basis for decision criteria for Tiered Aquatic Life Use (TALU) development.

5.1 The BCG as an Assessment Tool

The conceptual model of the BCG, as developed in Davies and Jackson (2006), incorporated ecological theory as well as widespread empirical experience of working aquatic ecologists. Development of an index that reflects the BCG required quantitative mapping of biological information into the conceptual and theoretical model. The mapping, or calibration, process of the index is simultaneously quantitative, empirical, and conceptual.

- The BCG is calibrated using a data set, but also requires ecological considerations with wide expert agreement. The result is intended to be more general than a regression analysis of biological response to stressors.

- The BCG uses universal attributes (attributes I to VI) that are intended to apply in all regions. Specifics of the attributes (taxon membership, attribute levels indicating good, fair, poor, etc.) do vary across regions and stream types, but the attributes themselves and their importance are consistent.

- The BCG requires descriptions of the classes or levels, from pristine to degraded. Although this requires extra work at the outset, it ensures that future information and discoveries can be related back to the baseline level descriptions. Level descriptions are not perfect or static—they will be altered by increases in knowledge.

The BCG may be more robust than current indexes because it allows for nonlinear responses, as well as having requirements for combinations of metric values in the condition classes. Also, the it is not conceptually tied to “best available” sites as an unalterable benchmark. Although best available sites are used as a practical ground truth, it is recognized at the outset that these sites are typically less than pristine, and may be a lower level (e.g., BCG levels 2, 3, 4).

5.2 The BCG and Aquatic Life Use

The terms “Use”, “Designated Uses”, and “Aquatic Life Use” have specific meanings for water quality management in the context of the Clean Water Act. A state defines the uses for its waters, and develops physical, chemical and biological criteria to protect those uses. Minnesota’s Tiered Aquatic Life Uses (TALUs) are aquatic life uses that are matched more closely to the Designated Uses, rather than a single one-size-fits-all aquatic life use (USEPA, 2005). The BCG, as a universal yardstick, is intended to be used in setting biological criteria to match specific TALUs. It is important to note that levels of the BCG are NOT equivalent to TALUs, although a given TALU level may be set to a level of the BCG. The BCG is a scientific measurement yardstick only; it does not express policy decisions and breakpoints for designated uses.

Designated Uses are intended to be set at the highest attainable use for a water body, taking into account natural limitations or irreversible physical (infrastructure) alterations to the habitat or watershed (e.g., existing urban infrastructure, flood control, harbor facilities, irrigation, etc.). Infrastructure is not always irreversible: roads can be modernized, many older dams and obstructions are being removed from streams, habitat can be restored, etc. Designated uses thus also include potential quality or condition that may not currently be attained, but could be attained with appropriate controls or restoration. Thus, Aquatic Life Uses can be set according to the biological potential of waterbodies, not according to their current condition.

The BCG provides a powerful approach for an operational monitoring and assessment program, for communicating resource condition to the public and for management decisions to protect or remediate water resources. The levels of the BCG are biologically recognizable stages in condition of stream waterbodies. As such, they can inform a biological basis for biological criteria and regulation of Minnesota's waterbodies. Adoption of the BCG as an assessment tool in the context of multiple Aquatic Life Uses (Tiered Uses) yields the technical tools for protecting Minnesota's highest quality waters, as well as developing realistic restoration goals for urban and agricultural waters. The BCG allows practical and operational implementation of multiple aquatic life uses in a state's water quality criteria and standards.

5.3 Technical Recommendations

We recommend the following:

- Test rules with new (unassessed) sites to determine model and panel concordance. As new data are added to Minnesota's biological database, panel assessments for a subset of these data should be performed to test the models to ensure that the models are broadly applicable to streams across the state. Identification of sites that do not fit the current BCG models can be used to refine these models to improve their performance. Expansion of the calibration dataset will reduce "over fitting" to the original dataset. This approach can also help to identify stream reaches that do not fit into the current stream classification framework and may need site-specific criteria or a new stream classification.
- The fish logic rules and model were the most troublesome to develop and readjust to the two headwaters categories: Northern Headwaters and Southern Headwaters. In part, this was due to the small number of species in these small streams, and the few fish species found in these habitats tended to be tolerant. This resulted in a limited assemblage of species and tolerances that make assessments problematic, both by the panel and the model. However, the models developed for these classes reasonably predicted panel decisions (77-88%). We recommend that the fish BCG for the two headwater stream classes be reviewed further to demonstrate that the BCG rules are consistent and reliable.

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Ecological Perspective on Water Quality Goals

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ABSTRACT / The central assumption of nonpoint source pollution control efforts in agricultural watersheds is that traditional erosion control programs are sufficient to insure high quality water resources. We outline the inadequacies of that assumption, especially as they relate to the goal of attaining ecological integrity. The declining biotic integrity of our water resources over the past two decades is not exclusively due to water quality (physical/chemical) degradation. Improvement in many aspects of the quality of our water resources must be approached with a much broader perspective than improvement of physical/chemical conditions. Other deficiencies in nonpoint pollution control programs are discussed and a new approach to the problem is outlined.

Increased societal concern for deteriorating water resources in the United States is clearly manifest in the passage of water pollution control legislation during the past decade. The Clean Water Act has set forth the objective of restoring and maintaining the "...chemical, physical, and biological integrity of the Nation's waters..." The Act further established a goal of eliminating the discharge of pollutants by 1985, and an interim goal of achieving, wherever attainable, water quality that provides for the protection and propagation of fish and provides for recreation in and on the water (commonly referred to as the "fishable/swimmable" goal). The major efforts and funding of water pollution control programs in the 1970s focused on point sources of pollution because it was relatively easy to control and regulate them. As the magnitude of the point source pollution problem was reduced, the relative contribution from nonpoint sources increased. Large scale efforts to curb nonpoint pollution are just beginning. However, before major expenditures are made, our experiences with several nonpoint pollution control projects prompts us to recommend an examination of water resource problems in the United States with respect to the current legislative framework.

The purpose of this paper is to examine the objectives and goals of the Clean Water Act and the ability of current programs to meet those objectives. We accomplish this through examining nonpoint pollution abatement programs in the United States and draw upon our experiences with an ongoing study, the Black Creek Project in Allen County, Indiana. We discuss the concept of biological integrity and outline briefly the fundamentals of stream biology to emphasize the need for a holistic approach to water resource management. Two alternatives are explored: (1) traditional soil and water conservation management, and (2) an innovative approach designed to restore biological integrity.

KEY WORDS Nonpoint source pollution, Water quality, Stream ecosystems, Flow regimes, Clean Water Act, Allen County, Indiana; Biotic integrity

The Black Creek Project

In 1972 the Allen County, Indiana, Soil and Water Conservation District, with assistance from the USDA-Soil Conservation Service, Purdue University, and the University of Illinois, began investigating nonpoint source pollution in a 48.5 km^2 (12,000 acre) subwatershed of the Maumee River basin under a grant from the U.S. Environmental Protection Agency. This study, commonly called the Black Creek Project, was the first detailed look in the United States at the contributions of agriculture to the degradation of water quality and ultimately to a reduction of environmental quality. The Black Creek Project, although now providing information of use to Section 208 planners, actually predates the adoption of Public Law 92-500 which, in part, requires an analysis of the impact of nonpoint source pollution on water quality. It was funded under provisions of the 1969 Water Quality Act calling for special demonstration projects to improve the quality of water in the Great Lakes (Morrison 1977a).

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The Soil Conservation Service supplied technical assistance for implementing traditional soil and water conservation practices within the study watershed. Over a five year period, a sum of $519,000 was spent on cost-sharable practices and, although not 100 percent successful, the project was able to implement a great number of soil and water conservation practices. Researchers from Purdue University and the University of Illinois, representing agricultural engineering, agronomy, agricultural economics, rural sociology, and the biological sciences, investigated the impact of the project on various components of the system. Detailed results are reported by Morrison (1977b). The discussion that follows is an outgrowth of our experiences with the Black Creek Project and similar nonpoint pollution abatement programs in Indiana, Illinois, and Ohio.

Biological Integrity and the Fishable/Swimmable Goal

Is the objective of chemical, physical, and biological integrity equivalent to the fishable/swimmable goal? Although these terms were not precisely defined in the Clean Water Act, it is clear that the two concepts are not equivalent. The interim goal is to achieve a level of *water quality* that is compatible with fishing and swimming in a waterway. Water quality is traditionally interpreted as the physical/chemical properties of water, a fact that greatly limits the scope of the goal. We believe other factors (discussed below) that may affect the actual attainment of fishable/swimmable conditions are not adequately addressed by existing pollution control and water resource management programs. For example, water quality standards for physical/chemical parameters have served as surrogates for the fishable/swimmable goal. States set water quality standards (WQS) based upon the criteria necessary to protect aquatic life and human health and, thus, compliance with WQS implies the attainment of fishable/swimmable waters. A comprehensive evaluation of both physical/chemical and biological data is a better determination of whether or not fishable/swimmable conditions are being achieved.

The concept of integrity mentioned in the Clean Water Act is, at best, elusive. A comprehensive symposium sponsored by the Office of Water and Hazardous Materials of USEPA (Ballentine and Guarraie 1975) did not produce a clear definition of integrity but several contributors strongly urged that the water resources of the nation be considered from a holistic (systems) perspective. Thus, unlike the fishable/swimmable goal, the integrity objective encompasses all factors affecting the ecosystem and can be defined as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region." A similar definition of ecological integrity was given by Cairns (1975). The summation of chemical, physical and biological integrity can be equated with ecological integrity. A system possessing integrity can withstand, and recover from, most perturbations imposed by natural environmental processes, as well as many major disruptions induced by man. A thoughtful discussion of these ecological concepts, including their measurement and management applications, was provided by Westman (1978).

Note that this definition does not make specific mention of resource value in terms of man's use of water (beneficial uses). Instead, there is an implicit recognition that a functioning ecological system is the ultimate resource upon which man depends. As Woodwell (1975) has stated: "These are the resources that are used by all of the people on earth, all of the time." Only in the presence of a functioning biological system are other resources (for example, energy, minerals) useful to man. Some would argue that it is unrealistic to maintain ecosystem integrity as defined above. To these individuals, the beneficial uses of water are of greater and more immediate concern to the continued functioning of our society. Granted, it is unrealistic to adhere to the goal of fully natural ecosystems at the expense of beneficial resource use, but it is also unrealistic to assume that our environment can continue to absorb an accumulation of intrusions on the integrity of the biosphere (Woodwell 1975). A middle ground is required and was, we believe, the intent of the Congress when it enacted the Clean Water Act.

A compartmentalized model developed by Odum (1969) is useful in visualizing the middle ground we are seeking. It is a simple representation of the basic functional types of environments required by man (Fig. 1): 1) productive environments, for example, agriculture; 2) protective environments, for example, natural areas preserving ecological integrity; 3) a compromise between 1 and 2; and 4) urban-industrial environments. As will be discussed below, the compartment model is useful for addressing, in operational terms, strategies of innovative soil and water conservation management.

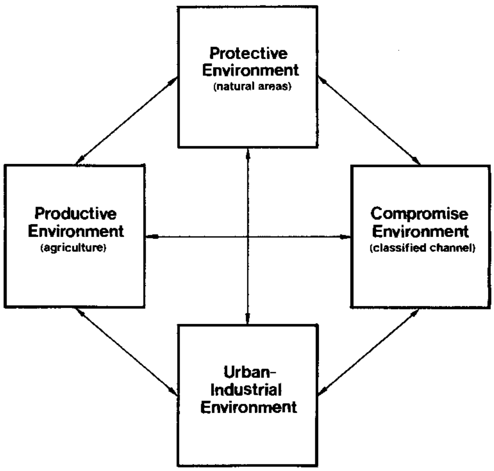

Figure 1. Compartment model of the basic kinds of environment required by man, partitioned according to ecosystem development and life-cycle resource criteria (modified from Odum 1969).

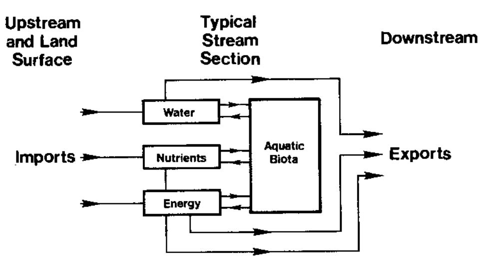

Figure 2. Generalized flow diagram for an aquatic ecosystem.

Stream Ecosystems

An individual stream or section of a stream is not an isolated system. Streams and rivers are open ecosystems with dynamic imports and exports of nutrients, energy, and water (Fig. 2). Inputs to upstream (headwater) areas are ultimately carried to and affect downstream areas (Meade and Trimble 1974). But movements are not limited to transport from upstream to downstream areas. Many aquatic organisms, especially fishes, may depend upon migration among stream reaches for the completion of their life cycles. Thus, stream ecosystems must be considered in terms of extensive geographic areas and as dynamic, open ecosystems.

This ecological reality illustrates the weakness of using the interim fishable/swimmable goal as a terminal objective. Fishable, in this context, is often defined as making the stream useful to fishermen in capturing sport or commercial fish. However, since many small streams contain too little water to be used for swimming or to support a sport or commercial fishery, they are often discounted as not having any significance to the fishable and swimmable objective. We feel that it is inappropriate to measure the value of a stream reach based on this particular component of fishable and swimmable criteria. That quality must be more broadly defined than hook-and-line locally because the importance of headwater streams to downstream reaches (in terms of production of fishable benefits downstream) is underemphasized in that context. Although a headwater stream may never be fishable, it is an integral component of the watershed; its preservation is essential if downstream reaches are to be fishable and swimmable (Karr and Dudley 1978). The biological integrity mandate of the Clean Water Act depends on an overview of the entire water resource system at the watershed level rather than isolated consideration of local stream reaches.

The concept of the open ecosystem has two other implications. First, streams are subject to rapid and gross perturbations caused by land-use changes (urbanization, intensive agriculture). Secondly, properly managed land use in watersheds can effectively and rapidly lessen perturbations in stream systems.

A classification system developed by Horton (1945) and modified by Keuhne (1962) is commonly used by aquatic biologists to discuss the progressive increase in stream size. According to this system, the smallest streams in a watershed are first order. When two first-order streams join, they form a second order stream; when two second-order streams join, they form a third order stream. Ecological discussions of streams typically consider three size classes: the headwaters (1st to 3rd order), intermediate-sized rivers (4th to 6th order), and large rivers (7th and larger orders). While this classification system is generally useful, note that stream order effects may vary somewhat among watersheds. For example, differences in size of upstream watershed or watershed topography may affect the nature of the stream-order pattern.

Man alters streams by dredging new channels in poorly drained areas or by modifying existing natural channels.

These man-engineered watercourses must be considered streams even though they are clearly different from natural streams in many respects (for example, drainage and flow characteristics, chemical and physical conditions, bottom type). Important as these differences are, one basic ecological principle applies to both man-altered and natural streams; water, nutrients, and energy are exported to downstream areas. Thus, man's construction of drainage ditches is not separate from natural drainage patterns; rather, it is only an addition to or a modification of the natural stream network that profoundly affects water resources both locally and downstream.

We have been able to identify what we feel are four major classes of variables (Fig. 3) which, when modified by man's activities, play primary roles in determining the ecological integrity of running water (lotic) ecosystems (Karr and Dudley 1978). These are flow regime, water quality, habitat structure, and energy source.

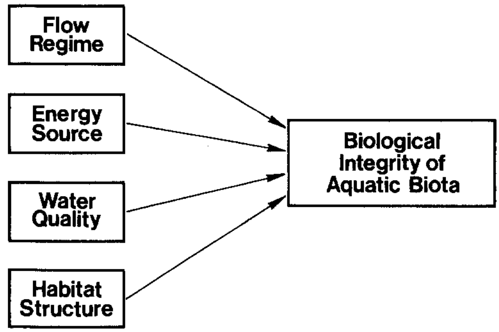

Figure 3. Primary variables affecting the structural and functional integrity of the biota of a headwater stream (modified from Karr and Dudley 1978).

Flow Regime

Fluctuating water levels are an integral part of all stream ecosystems and aquatic organisms have evolved to compensate for changing flow regimes. Even areas decimated by catastrophic floods or droughts are often quickly recolonized (Larimore and Smith 1963, Horwitz 1978). But modifications of the land surface with changing land use typically result in flood peaks and low-flow periods that are more severe as well as more frequent. Late summer low-flow periods may be extended while hydrograph peaks following runoff events are often of shorter duration.

High water periods are determined by the frequency, occurrence, and type of rainfall event, the timing of those rainfall events, and such antecedent conditions as soil moisture, time since the last rain, and amount and type of soil cover. Flood events in natural watersheds tend to have a dampened hydrograph, while those in agricultural watersheds tend to have a sharp and extreme peak (Bormann and others 1969). Low flows in natural watersheds tend to be severe only in particularly dry years, while low-flow periods in modified watersheds, especially those with extensive drainage systems, are relatively more severe, especially during late summer and early fall periods when rainfall is at relatively lower levels in midwestern portions of the United States.

When such flow events prevent seasonal migrations of fish or interfere with egg or fry development, irreversible catastrophic changes may result. Under the extreme condition of dewatering, the biota may be lost entirely. Recognition of the significance of this problem has precipitated the formation of a special group within the Office of Biological Services of the U.S. Fish and Wildlife Service. This group, the Cooperative Instream Flow Service Group, is developing a detailed methodology for evaluating flow requirements of aquatic organisms. The primary objective is to develop criteria to assess the impact of altered stream-flow on habitat characteristics and the use of an area by aquatic organisms (Stalnaker and Arnette 1976). Efforts are underway to identify the hydraulic conditions necessary for a variety of organisms, including different age classes of the same species. For example, the distribution of walleye as a function of flow is given as a probability distribution that varies among the age classes and with the reproductive state of fish (Fig. 4). Fry are found in only the slowest water, while juveniles, and especially adults, utilize higher velocities. Finally, spawning fish require much higher flow rates. Modifications in a stream that destroy areas with "spawning" velocities may have a significant negative effect on walleye reproduction even though adult fish may not be directly affected. These efforts that examine the flow regimes and hydraulics of streams and their effects on ecological integrity will make major contributions to the management of running water resources.

Water Quality

In recent years most efforts to reverse the degradation in the quality of water resources have focused on the physical and chemical properties of water. Temperature, dissolved oxygen, concentrations of soluble and insoluble organics and inorganics, heavy metals, and a wide variety of toxic substances are components of special interest.

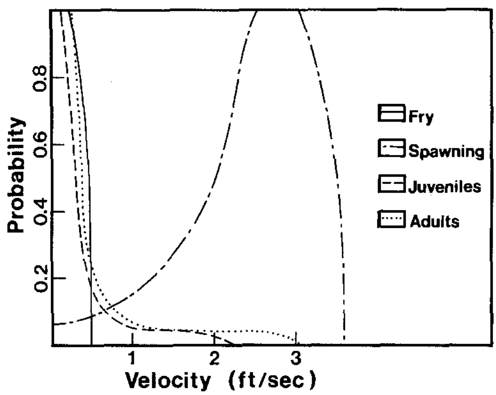

Figure 4. Probability of use curve for several age classes of walleye (*Stizostedion vitreum*) (adapted from unpublished material of the Cooperative Instream Flow Service Group, with permission of C. Stalnaker).

They may affect biological integrity by directly causing mortality or may shift the balance among species as a result of subtle effects such as reduced reproductive rates or changing competitive ability.

The importance of these factors on stream biota is widely known (Warren 1971, Hynes 1974). Water quality factors that are of special concern include light, temperature, dissolved oxygen, suspended solids, dissolved ions and other materials. These play critical roles in determining an area's suitability for aquatic organisms. In addition to the average conditions, extremes and their temporal patterns have important impacts on biota.

Each of these is of concern. In many watersheds, human activities may precipitate the degradation of ecological integrity because of the synergistic effects of several variables (see discussion of algal blooms below).

Habitat Structure

The physical structure of the environment also plays a major role in determining the number and kinds of fishes and other organisms that can survive in a stream. Channel geometry in natural watersheds is typically meandering, with substrate diversities created by varying flow regimes length-wise and across the channel. The result is substrate sorting, the presence of pools and riffles, erosion and deposition areas, and ultimately a dynamic equilibrium between the flowing water and its substrate. In contrast, stream alterations, such as channelization, produce channels with little pool and riffle development and uniform substrates and depth. In addition, sedimentation increases as a result of a disequilibrium in a channel and/or because of erosion from the land surface. Finally, straight, open channels in the presence of abundant nutrients, sunlight, and high temperatures create ideal conditions for algal blooms. In years of below-normal rainfall in late spring or early summer, these algal blooms develop in late May and early June; in years with more substantial rainfall during the early summer, the algal blooms are curbed by the flushing action of channel flow.

These and other complex interactions with the physical habitat of streams affect the biota of the stream. Bottom-dwelling invertebrates such as mollusks (Harman 1972) and insects (Allan 1975) seem to be especially affected by the diversity and sorting of bottom or substrate types in an area (sand, gravel, rocks, etc.). Substrate particle size determines the size of the interstitial spaces which, in turn, affects the type of bottom-dwelling community. Adequate interstitial space is essential for the movement and feeding of many aquatic invertebrates. Fishes, which use environments in a more three dimensional fashion, seem to respond to a complex of structural features including substrate type, depth, and current velocity (Gorman and Karr 1978). Further, many fishes and some invertebrates require places of concealment (cover) as feeding locales or as places to escape predation. General cover types include undercut banks, timber and brush snags, and aquatic vascular plants. Without essential habitat structure, many forms of aquatic life are eliminated from streams. If we measure habitat diversity as a mosaic of depth, current and substrate conditions, and fish species diversity (both using the Shannon's index; see Gorman and Karr 1978 for details), it is clear that more diverse habitats support a greater fish species diversity (Fig. 5). Thus, nonpoint control efforts that produce high water quality (physical/chemical conditions) may fail to produce a water resource with high biotic integrity if suitable physical habitats are absent.

Two recent research efforts illustrate the importance of considering habitat structure as a primary determinant of the quality of a water resource. In one case, we detected considerable movements by fish in a number of regions in a study watershed (Black Creek) in northeast Indiana. To study these movements, we marked fish with a procedure called cold branding. Silver brands with the shapes of various letters were supercooled with liquid nitrogen and touched to the sides of fish, duplicating the common hot branding used to mark livestck on open

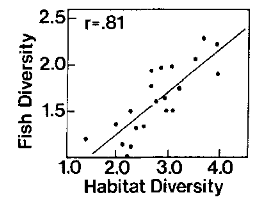

Figure 5. Relationship between habitat diversity and fish species diversity (from Gorman and Karr 1978).

ranges. We branded fish in three major habitat types. Three sampling stations were selected in the main channel of Black Creek in areas that had been subjected to major channel alterations early in the study. The second major habitat was on the Wann Drainage immediately east of the Black Creek watershed. Although there has been no recent channel modification work in this area, the stream reach had been modified approximately ten years earlier. The lack of disturbance over the years created a stream that had begun to meander in its channel base and in which dense vascular plant populations provided cover. As reported earlier by Gorman and Karr (1978), this section of stream contained a richer fauna than that found in similar reaches of the Black Creek watershed. The third study area was in the Black Creek watershed on the Wertz Drain where it traversed a woodlot and had an especially rich fauna (Gorman and Karr 1978).

Populations in higher quality habitat are relatively more secure (Table 1); they are able to survive locally over longer periods. Clearly, total emphasis on water quality in the physical/chemical sense will not overcome habitat structure deficiencies. Further, we have provided evidence in earlier reports that those areas with better quality habitat also have a beneficial effect on water quality (Karr and Gorman 1975; Karr and Schlosser 1977, 1978; Schlosser and Karr 1980).

In another study, one member of the group at the University of Illinois divided two sections of Jordan Creek in east-central Illinois with 1/4-inch mesh hardware cloth supported by steel posts. On one side of each section all cover features (logs, limbs) were removed from in or near the water. On the other side, a continuous series of similar objects was secured along the stream. In July and September, samples of the biomass of fish were 4.8 to 9.4 times as high in the areas with structurally complex habitats. Further, the larger fish, and especially the top predators, tended to select the structured habitat. In this case we know that water quality is the same in the structured and unstructured sides of the stream, yet the numbers of fish are markedly different. These improved habitat conditions seem to provide two things: habitat for small fish including a diversity of substrates for food organisms, and hiding places (cover) from which large fish can prey on smaller species. This emphasizes the importance of habitat structure as a determinant of biotic conditions in a stream.

Energy Source

In stream ecosystems, the form and source of the energy and nutrients are especially important in determining ecosystem characteristics. The energy contained in the chemical bonds of organic matter is the basic energy source for animals, fungi, and many bacteria. The process of breaking the chemical bonds to release energy and simpler compounds is respiration. Production is the reverse process in which energy, in the form of solar

Table 1. Recapture rates, habitat diversity, and stream channel conditions at several sites where fish were marked by cold branding

Stream	Channel and habitat conditions	Habitat diversity*	Number of fish marked	Fish recaptured (%)
Black Creek	Badly disturbed	2.89	1,190	5
Wann Creek	Disturbed, but recovering	3.05	767	15
Wertz Drain in Wertz Woods	Relatively natural	3.31	958	37

*This is an index of complexity of stream habitat as a composite of substrate types (sand, pebble, rock, etc.), water depth, and current velocity using the information theoretical measure of diversity. Higher values indicate stream habitats of greater complexity. See Gorman and Karr (1978) for a detailed explanation of methods.

Table 2. General characteristics of running water ecosystems according to size of stream (modified from Cummins 1975)

Stream size	Primary energy source	Production (trophic) state*	Light and temperature regimes	Trophic status of dominant	
				Insects	Fish
Small headwater streams (stream order 1–3)	Coarse particulate organic matter (CPOM) from the terrestrial environment Little primary production	Heterotrophic $P/R < 1$	Heavily shaded Stable temperatures	Shredders Collectors	Invertivores
Medium-sized streams (4–6)	Fine particulate organic matter (FPOM), mostly Considerable primary production	Autotrophic $P/R > 1$	Little shading Daily temperature variation high	Collectors Scrapers (grazers)	Invertivores Piscivores
Large rivers (7–12)	FPOM from upstream	Heterotrophic $P/R < 1$	Little shading Stable temperatures	Planktonic collectors	Planktivores

*A stream is autotrophic if instream photosynthesis exceeds the respiratory requirement of organisms living in the area (that is, $P/R > 1$). It is heterotrophic if importation of organic material from upstream areas or the land surface is necessary (that is, $P/R < 1$).

radiation and simple compounds, is converted into complex organic compounds. Obviously, plants are the major producer organisms and high production rates are dependent upon abundant sunlight and essential nutrients. The fundamental energy relationship can be expressed by the production (P) to respiration (R) ratio: $P/R > 1$ when production exceeds respiration (autotrophy); $P/R < 1$ when respiration exceeds production (heterotrophy). In streams, this basic energy flow characteristic is sensitive to the organic loading from the terrestrial environment, the amount of sunlight and nutrients, the form or availability of nutrients (simple compounds vs. complex organic compounds), and a number of other factors such as turbidity.

Studies of the energetics of stream ecosystems (Cummins 1974) stress process-oriented attributes such as production, respiration, energy flow, nutrient cycling, and trophic dynamics. It is a fundamental postulate that many process-oriented attributes of running water ecosystems change as streams increase in size from headwaters to mouth.

The transition from small headwater areas to major rivers is referred to as the stream continuum. Structural and functional attributes of natural stream ecosystems change along this continuum (Table 2). These attributes serve as reference points for assessment of the status of the stream ecosystem in any location. If the ecosystem in a region differs from these expectations, the difference may be due to ecosystem degradation resulting from man's activities. At the very least, it suggests that more detailed study is required. The theoretical foundation for these "reference points" comes to a great extent from forested watersheds. As a result, it may be necessary to develop an alternate foundation for markedly different terrestrial environments in the dry nonforested regions of western North America (Minshall 1978).

Headwater streams in natural watersheds of eastern North America are usually heterotrophic. That is, they have production to respiration ratios (P/R) of less than 1.0 and are dependent on food produced outside the stream (allocthonous material). Dense tree canopies shade the headwaters so that instream production is minor, generally from small populations of moss or periphytic algae (algae attached to rocks or other substrates). One study in a New Hampshire watershed (deciduous forest) showed that 99 percent of the energy requirements for the biota of a headwater stream were of allocthonous origin (Fisher and Likens 1973). A very different watershed in Oregon (coniferous forest) demonstrated the same general pattern (Sedell and others 1973). In this situation the persistence of the biotic community depends on a regular input of food (organic matter) from external sources. The terrestrial environment supplies much of the energy input in the form of leaf litter shed in a predictable seasonal pattern (fall in temperate deciduous forest, dry season in tropical forest).

The particle size of organic matter entering a stream is just as importnt to stream ecosystem functioning as the amount, type, or timing of energy input. In undisturbed headwater areas, the terrestrial environment produces particulates of relatively large size (such as leaves, and twigs), referred to as coarse particulate organic matter (CPOM). Bacteria and fungi quickly colonize the CPOM and, as a result of their metabolic activity, speed the process of fragmentation into smaller particles—fine particulate organic matter (FPOM). (Any organic particle less than 1 millimeter in diameter is considered FPOM, regardless of its source.) The breakdown process of CPOM is accelerated by benthic invertebrates, primarily aquatic insects, which ingest and further fragment (or shred) the CPOM. Organisms with this functional capacity are called shredders. Shredders utilize some of the energy contained in the CPOM along with the rich growths of attached bacteria and fungi. But most of the CPOM is simply converted to FPOM and is available for use by another functional group of aquatic organisms called collectors.

Collectors either filter FPOM from the water or gather it from the sediments (Cummins 1973). Because of structural adaptations, most collector organisms utilize FPOM only within a narrow size range (Cummins 1974), thus illustrating the critical nature of particle size in stream ecosystems. The natural association of shredder and collector organisms in headwater streams results in a highly efficient utilization of energy (organic matter) input. Cummins (1975) has estimated that the biota process about 80 percent of the particulate organic matter (POM) and 50 percent of the dissolved organic matter (DOM) in natural first to third order streams.

Functional attributes are markedly different in undisturbed intermediate-sized rivers. The stream becomes autotrophic ($P/R > 1$) as the stream becomes less shaded and algae and vascular plants increase in abundance. CPOM inputs are reduced, resulting in decreased shredder abundance. Incoming allocthonous material is primarily FPOM from headwater areas and a variety of collector organisms is common. The autotrophic status of the stream accounts for the presence of a third functional group of aquatic macroinvertebrates. These are the scraper or grazer organisms that exploit periphytic algae and vascular plants. A few scrapers can always be found in natural headwater streams, but their abundance is severely limited by the low rate of primary production.

In large rivers (7th to 12th order), the stream again becomes heterotrophic primarily because of increased turbidities reducing light penetration and, therefore, the potential for photosynthesis (Cummins 1973). The primary production that does occur is generated by phytoplankton (free-floating algae). Free-floating collectors (zooplankton) are also present, utilizing the phytoplankton and suspended FPOM as food. Collectors also predominate in the sediments, as FPOM is the major energy source. Few scapers or shredders occur in a large river environment.

The fish fauna also reflects the energy sources available in a stream. However, fish can be more directly related to the value of the water resource (commercial or sport fish) in human terms. Cummins (1975) categorized the functional attributes of fish communities according to the food habits of the dominant fish. Predominant food habits are somewhat different for the three major ecological areas of an undisturbed river system. In headwater streams, fishes that feed upon macroinvertebrates (invertivores) dominate. Invertivores along with piscivores (fish that consume other fish) dominate intermediate-sized rivers. Finally, in large rivers, dominant members of the fish community are planktivores (fishes feeding upon both phytoplankton and zooplankton). Two additional categories are omnivores (consuming both plant and animal matter in approximately equal portions) and herbivores (consuming primarily plant materials). Omnivores and herbivores are rarely dominant in natural running water systems.

Our experience in modified and natural watersheds in Indiana, Illinois, and Iowa indicates major disturbances in these energy source (functional) dynamics. Many modified headwater areas seem to be more autotrophic than heterotrophic (Table 3) because of the abundance of sunlight and nutrients. Algal blooms alter the organic load and habitat characteristics of the stream. This, in turn, affects the aquatic invertebrate community, organic matter processing, and, thus, organic loadings downstream. In addition, there is some evidence that the trophic status of fishes shifts from piscivores to omnivores because of declining water quality, resource base,

Table 3. General characteristics of natural (Cummins 1974) and modified (Karr and Dudley 1978) headwater streams in eastern United States

Parameter of interest	Natural	Modified
Water quality		
Light and temperature	Heavily shaded Stable temperatures	Open to sunlight Very high summer temperature
Dissolved oxygen	Relatively stable	Highly variable
Suspended solids concentration	Low to very low	Highly variable
Dissolved ions	Generally low	High, especially for *P* and *N*
Flow regime		
Flood events	Dampened hydrograph	Hydrograph peaks sharp and severe
Low flows	Moderately severe only in dry years	Moderately severe each year in late summer and early fall; extremely severe in dry years
Habitat structure		
Pools and riffles Meandering topography	Channel topography and substrate diversity in equilibrium with stream hydraulics	Reduced and/or destroyed by channel maintenance activities
Sedimentation	Minor except in a few unstable bank areas	Major problem with sediment source from land and from unstable banks; sedimentation decreases habitat diversity and directly abrades organisms
Energetics		
Particulate organic matter size and source	Predominantly coarse particulate organic matter from forested terrestrial environment	Less coarse and more fine particulate organic matter from agricultural and domestic sewage
Production (trophic) state	Little primary production Heterotrophic; $P/R < 1$	Algal blooms common Autotrophic; $P/R > 1$
Trophic status of dominant		
Insects	Shredders, collectors	Scrapers, collectors
Fishes	Invertivores	Invertivores but forced to select a broader range of food types
Migrant fishes	Top predators	Mostly filter feeders and/or omnivores

and habitat conditions (Karr and Dudley 1978). As a result, populations of less desirable fishes increase while top predator populations, which act as a natural population check on other species, decline.

In summary, then, we suggest that the attainment of ecological integrity in our water resource systems demands a broad conceptual approach. Several key problems to be addrssed in agricultural watersheds are reiterated here:

1. *Allocthonous organic matter inputs:* FPOM input from sewage and stormwater runoff is substantial, as evidenced by high bacterial contamination (Dudley and Karr 1979). This change, along with the modification in form and content of CPOM discussed earlier, results in major structural and functional changes in the stream ecosystem.
2. *Nutrient availability:* Concentrations of simple nutrient forms (PO_4, NO_3, NH_4) do not limit algal populations. In addition, inputs of complex organic compounds associated with CPOM are not effectively processed.
3. *Sunlight availability:* A predominance of unshaded stream channels results in high solar energy input.

Coupled with available nutrients (#2 above), this results in buildup in algal populations (CPOM), which are either subject to slow decay in the headwaters or are washed downstream in large quantities during high flows. These algal blooms add to the organic load of the aquatic system and change the physical characteristics of the stream environment (reducing current velocities, covering natural substrates).

4. *Temperature and dissolved oxygen imbalance:* Seasonal and daily patterns of temperature and dissolved oxygen are exaggerated and poorly buffered from environmental influences (weather extremes, organic loading).
5. *Stream habitat characteristics:* The diversity and stability of high quality stream habitat are low (Gorman and Karr 1978). The ditching and drainage efforts prevalent in many agricultural watersheds perpetuate this problem.
6. *Seasonal low flows:* The loss of natural vegetation and installation of complex drainage networks results in rapid runoff instead of slow release of excess water. As a result, extreme low flows during dry periods, especially in late summer and early fall, place considerable stress on aquatic ecosystems.
7. *Changes in insect and fish communities:* These and other shifts in the four primary variables (individually and in the aggregate) cause major shifts in benthic insect faunas as well as the fish communities. In addition, because of the effect of these changes on the use of headwaters as spawning and nursery areas, the fish of downstream areas are also affected (Karr and Dudley 1978).

Water Resource Management in Agricultural Watersheds

The central assumption of most agricultural nonpoint pollution control programs has been: traditional soil and water conservation practices are sufficient not only to reduce erosion and other nonpoint pollutants but also to improve the quality of the water resource. That is, it is possible to manage water resource problems resulting from agricultural land use through a voluntary soil and water conservation program. Numerous demonstration projects as well as the proposed Rural Clean Water Program are employing this basic assumption. We now examine the ability of existing programs (alternative A) to meet the goals and objectives of the Clean Water Act in contrast to the ability of management programs that incorporate the principles of stream ecosystems (alternative B).

Alternative A. Traditional Soil and Water Conservation Management

The typical nonpoint pollution control program concentrates on a list of erosion control and animal waste control practices used by the Soil Conservation Service. This list is then reduced to a subset of Best Management Practices (BMPs) thought to have some value in improving water quality. The disadvantage of this approach is that a number of other activities that may result in improvements in the quality of the water resource are not considered. Further, the potential benefits of an integrated network of erosion control practices, coupled with practices that may only benefit water quality, may be greater than the benefits from erosion control practices alone.

Traditional soil and water conservation management does not effectively consider the principle of ecological integrity. The primary focus of the management agencies involved (Soil and Water Conservation Districts, Soil Conservation Service, Agricultural Extension Service, and Agricultural Crop Stabilization Service) is maintaining agricultural productivity through erosion control, land drainage, and the management of soil fertility (Carter 1977, Morrison 1977c). Thus, cropland is the unit being managed with benefits going to both cropland and, presumably, downstream waterways. Water resource benefits occur in downstream waterways because of reduced pollutant loading. Lakes and large rivers in highly agriculturalized areas would be expected to receive the greatest benefits from reduced pollutant loading. Traditional soil and water conservation programs are clearly needed in waters where sediment, nutrients, or toxics are a problem.

However, the major shortcoming of the BMP approach is the failure to consider the stream ecosystem between the cropland and the downstream waterway where the benefits of reduced pollutant loading show up. These stream ecosystems are headwaters and intermediate-sized rivers that have often been drastically disturbed by agricultural land-use practices. If reduced pollutant loading has any beneficial impact on these small streams and rivers, it is imperceptible due to major perturbations in flow regime, habitat structure, and energy dynamics. In summary, the effectiveness of BMPs in achieving water quality compatible with fishable/swimmable conditions has not been proven; and neither are BMPs geared towards reaching the objective of ecological integrity.

Alternative B. Innovative Management to Restore Ecological Integrity

Soil conservation practices (BMPs) applied to the land have water quality benefits but they are only a part of a system of practices required for the sound management of water resources, including stream ecosystems. The time is right for careful application of an expanded list of BMPs into BEST MANAGEMENT SYSTEMS (Karr and Schlosser 1978). The following questions must be routinely asked: What will be the effect of the juxtaposition of several practices? How will they affect the widest range of water resource characteristics, not just how will they affect erosion control on the land, or water quality? What are the impacts of these on ecological integrity?

We must regularly examine the impact of nonpoint activities with and without varieties of management alternatives. It is important that the assessment include both local and downstream areas, as well as upstream areas. A further advantage of planning for integrated best management systems is that they may allow society to capitalize on the benefits to water quality that may accrue from the presence of integrated biotic communities. We may be able to capitalize on the ability of biota to serve as a natural treatment facility, rather than depending upon technological capabilities to improve water quality. Those technological capabilities often have higher societal costs than natural systems (Karr and Schlosser 1978).

The foundation of innovative management to restore ecological integrity is a conceptual model of an integrated land-use program based on Odum's model of environments required by man (Fig. 1). Man clearly needs productive (that is, agricultural) environments. However, protective environments that preserve biological integrity are also needed in all ecosystems to insure their continued functioning. If streams and rivers in highly agriculturalized areas are to be included in the national mandate for ecological integrity, then we believe it is necessary to incorporate the sound management of type 3 environments within those river ecosystems. The type 3 environment represents a compromise between productive and protective uses.

Many systems of land management might be applied to the Black Creek watershed in an effort to optimize production (agriculture) and protection (ecosystem integrity) through the designation of type 3 environments. Farming need not be eliminated from these areas; substitution of alternatives to continuous row crops such as rotation with limited row crops, conservation tillage systems, improved pasture management with the elimination of woodlot grazing, and permanent vegetation cover on erosive slopes and along stream banks are possibilities. All of these practices are commonly used to reduce on-site erosion. In the case outlined here their use will also be valuable as they affect sediment delivery rates to the stream channel and, in addition, help to stabilize stream channels (Karr and Schlosser 1978). Two extremes of distribution of protective environments are proposed in Fig. 6. A wide diversity of intermediate alternatives could be developed to satisfy local needs. An intensive research program is necessary before informed decisions can be made on optimum management programs. A key issue to address is the percentage of type 3 environment needed for a given level of ecological integrity in a river basin.

The important concept is that the land and its associated biota play a primary role in regulating the quality of a water resource. In type 3 environments the management strategy is to effect improvements in the four variables that influence ecological integrity while keeping the impacts on the productive components of the environments at a minimum. Some specific water quality benefits expected under such land use and vegetative cover management have been discussed by Karr and Schlosser (1978). Table 4 outlines a generalized management system that we believe would improve the biological integrity of headwater streams in agricultural areas. Practices aimed at improving water quality must be implemented in both type 1 and type 3 environments. The recommended practices for improving flow regime, habitat structure, and energy source are limited in application to areas designated as type 3 environments (Fig. 6). It is important to note that every watershed is unique and that practices and impacts can vary considerably among watersheds, as they do when planners select practices for erosion reduction. We realize land managed in this manner may not always be economically competitive in the current agricultural system. Potential mechanisms to solve this problem are now enumerated.

Implementation Mechanisms

It is not the purpose of this paper to analyze incentive programs that might speed implementation of alternative B outlined above. However, we can make some general comments on incentives in hopes of stimulating detailed analysis of their costs and benefits.

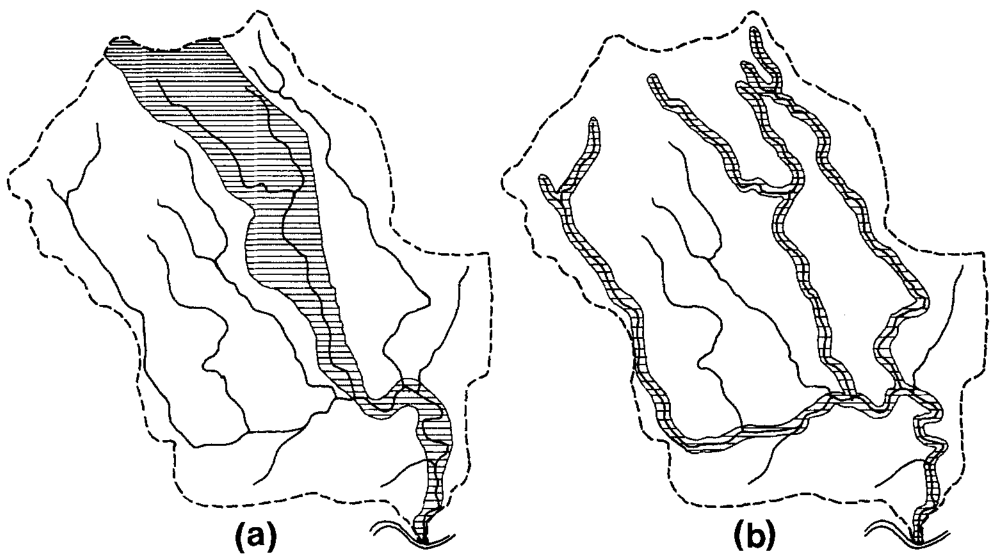

Figure 6. Black Creek watershed divided into type 1 (unshaded) and type 3 [shaded alternatives (a) and (b)] environments. Type 1 environments are productive and accommodate intensive agriculture. Type 3 environments represent a compromise between productive and protective qualities and function to preserve ecological integrity. Conservation practices in type 1 environments address water quality, while practices in type 3 environments address all four primary variables influencing ecological integrity. Alternatives (a) and (b) indicate two distributions of type 3 environments (classified streams) designed to improve water resources.

The objective of these and other incentives is to make less intensive farming on type 3 environments competitive with farming operations in type 1 environments while preserving some of the other environmental benefits of these areas. This can be accomplished by subsidies underwritten by society, the principal benefactor.

Classified Streams

The principle involved in setting aside areas for protection is well established. Unique natural areas or historical sites have long been protected from further development to enhance their long-term value to society. Federal agencies periodically implement set-aside programs to take land out of production or to conserve soil resources. An analog, a system of classified streams, should be developed to reduce local erosion and its effect on downstream water resources. Additional benefits from such programs might derive from increased availability of local recreational resources (Karr and Schlosser 1978). Since headwaters play an especially important role in determining resource quality throughout watersheds (Karr and Dudley 1978), efforts to benefit soil and water resources might emphasize a classified headwater approach.

Green Ticket

The basic outline of the "green ticket" program (Lake 1978) is to provide economic incentives to the farmer (or other land user) through governmental programs. These incentives must improve the profitability of a farm in exchange for installation and maintenance of needed conservation measures on the land. A sliding scale of incentives might exist to yield greater benefits to a farmer on areas identified as more critical. For example, areas that might be part of a larger classified headwater area might yield higher economic gain to the land owner than a patchwork of areas yielding lower benefit to society. We can even visualize groups of farmers exerting pressure on neighbors to develop a classified headwater program on their marginal land in the name of soil and water conservation benefit to society and economic benefit to them as individuals. Such programs should be encouraged on areas identified as locations where treatment of the smallest possible area (or at lowest economic cost) will yield the greatest benefit to society. Under these

Table 4. A generalized management system to improve the biological integrity of Black Creek and the anticipated impact on agricultural production within the watershed

Goal	Recommended practices	Impact on production
Water quality: reduction in sediment and nutrients	Traditional practices, especially conservation tillage, terraces, grass waterways, filter strips along stream channels, animal waste management plans, and soil fertility testing and management plans.	Production reduced slightly by conservation tillage on some soils; loss of cropland used for filter strips.
Flow regime: less extreme fluctuations in stream discharge	Augmentation of low flows through storage and later release of storm runoff and/or pumping ground water during dry periods. Conservation practices listed under water quality help in reducing peak stream discharge.	Minimal impact on production through augmenting low flows.
Habitat structure: improvements in stream habitat for fish and other aquatic life	Stream renovation (Nunnally 1978) practices instead of large scale streambank protection (channelization). Maximum preservation of natural habitat features (pools, riffles, meandering, cover, substrate size sorting, etc.).	The hydraulic improvements of channelization are only slightly greater than improvements under renovation practices. Agricultural production would not be affected by appreciably greater flood damages. In Black Creek, impaired tile drainage outlets are uncommon, meaning stream renovation would have little impact through the impairment of subsurface drainage.
Energy source: energy relationships capable of maitaining community structure and function	The management of a forested riparian environment that insures inputs of CPOM and a reduction in solar radiation. Additional water quality benefits such as improved temperature and dissolved oxygen and the trapping of sediment and nutrients are predicted under such management. An initial stocking of the stream with CPOM and aquatic invertebrates may be considered.	Loss of some cropland adjacent to streams.

circumstances, land holders might be eligible to collect extra Agricultural Crop Stabilization Service benefits, to pay lower rates on crop insurance, or to lower interest rates in federal loan programs. Further incentives could be integrated into local and state tax structures. Many other incentive programs could and should be sought. These must protect the economic state of the agricultural community and also produce the greatest benefit to society as a whole.

Summary

To conclude, we believe the results of experimental nonpoint pollution control efforts like the Black Creek Project demonstrate the need for improvement in the institutional approaches being taken to meet the goals of the Clean Water Act. Special concern must be placed on the attainment of ecological integrity rather than the interim goal of fishable and swimmable. Restoring and maintaining the quality of the nation's water resources require a new approach that encompasses the four primary variables of flow regime, habitat structure, water quality, and energy dynamics.

Acknowledgments

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THE INTEGRITY OF WATER

Proceedings of a Symposium

March 10-12, 1975

Washington, D.C.

U.S. Environmental Protection Agency
Office of Water and
Hazardous Materials

THE INTEGRITY OF WATER

a symposium

SYMPOSIUM COORDINATORS:
R. Kent Ballentine and Leonard J. Guarraia
Water Quality Criteria Staff, EPA
Washington, D.C.

OPENING SESSION

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Speakers: Kenneth M. Mackenthun
James L. Agee, Assistant Administrator, Office of Water and Hazardous Materials, EPA, Washington, D.C.
Thomas Jorling, Director, Center for Environmental Studies, Williamstown, Massachusetts
Donald Squires, Director, State University of New York Sea Grant Program, Albany, New York

CHEMICAL INTEGRITY

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Speakers: Bostwick Ketchum, Director, Woods Hole Oceanographic Institute, Woods Hole, Massachusetts
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INTEGRITY—AN INTERPRETATION

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Ronald B. Outen, National Resources Defense Council, Washington, D.C.
R. M. Billings, Director of Environmental Control, Kimberly-Clark, Neenah, Wisconsin
Gladwin Hill, National Environmental Correspondent, New York Times, New York

Following each presentation, Symposium participants were encouraged to question the speaker. These discussions were recorded by a professional reporting service and appear at the conclusion of each paper. They have been minimally edited, simply for clarification of the spoken word in print.

FOREWORD

"The Integrity of Water" results from the formal papers and comments presented at an invitational symposium by recognized water experts representing a variety of disciplines and societal interests. The focus of the symposium was on the definition and interpretation of water quality integrity as viewed and discussed by representatives of State governments, industry, academia, conservation and environmental groups, and others of the general public. The symposium was structured to address quantitative and qualitative characteristics of the physical, chemical, and biological properties of surface and ground waters.

It is recognized that streams, lakes, estuaries, and coastal marine waters vary in size and configuration, geologic features, and flow characteristics, and are influenced by climate and meteorological events, and the type and extent of human impact. The natural integrity of such waters may be determined partially by consulting historical records of water quality and species composition where available, by conducting ecological investigations of the area or of a comparable ecosystem, and through modeling studies that provide an estimation of the natural ecosystem based upon information available. Appropriate water quality criteria present quality goals that will provide for the protection of aquatic and associated wildlife, man and other users of water, and consumers of the aquatic life.

This volume adds another dimension to our recorded knowledge on water quality. It brings into sharp focus one of the basic issues associated with the protection and management of this Nation's valued aquatic resource. It highlights, once again, our unqualified dependence upon controlling water pollution if we are to continue to have a viable and complex society. The Congress has provided us with strong and comprehensive water pollution control laws. In accordance with the advances in research and development and with our increased knowledge about the environment, these laws will receive further congressional consideration and modification as appropriate. It is through the efforts of those who participated in making this volume possible that attention is focused once again on the basic goals of water quality to support the dynamic needs of this generation and of others to come.

Douglas M. Costle, Administrator
U.S. Environmental Protection Agency
June, 1977

CONTENTS

BIOLOGICAL INTEGRITY OF WATER— AN HISTORICAL APPROACH

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To a considerable extent the topics assigned to us have unreasonably artificial boundaries, because an ecologist cannot talk about the physics, chemistry, or biology of water separately, nor about the qualitative aspects of water and its biotas separately from the quantitative aspects. Moreover, in a meeting such as this of persons with disparate backgrounds and interests, effective communication can be a problem. We all tend to use words that have special meanings within our own disciplines and we assume a certain understanding of premises, principles, and laws when we use them. To make certain that we all are operating on the same wave length I shall present several principles of ecology that must guide our thinking about water, its management, and the potential effects on it of various manipulative processes, then give my own definition of the integrity of water, and finally address the topic assigned me. The principles to be discussed apply to all aquatic systems, but the examples I shall present will be concerned chiefly with lakes, as they, along with the oceans, provide in their sediments the only record of past events not covered by written observations or the memory of persons still living.

(1) Lakes and rivers are integral parts of larger systems—the watersheds or catchment areas that are drained by the rivers or drain through the lakes. Besides water itself, the catchment area contributes dissolved and particulate substances, both mineral and organic. In addition, usually lesser quantities of various substances are contributed directly to the water from the atmosphere by precipitation and dry fallout. Together with such process-regulating variables as light, temperature, current velocity, et cetera, these various substances comprise the abiotic portion of the aquatic environment and help control the diversity and abundance of aquatic organisms.

(2) Those substances that are used directly by aquatic organisms and are necessary in their metabolism—usually called essential nutrients—are recycled in the system by biological mechanisms. Storage in living biomass, in wood or sediments, or in the deep water of a stratified lake can delay the reutilization of these nutrients for varying periods of time. Because inputs and outputs, including storage, are generally in balance, an aquatic system to remain functional requires a continuous input of nutrients. The quantities of nutrients and other substances contributed by a watershed vary with the geological nature of the substrate and its overlying soils, the vegetational cover of the land, and climate. Since all of these tend to form regional patterns, it is not surprising that rivers and lakes also tend to form regional patterns or clusters in their chemistry, productivity, and biotic diversity.

(3) Besides nutrients there must also be a source of fixed energy, mostly in organic compounds. The latter derive both from photosynthesis accomplished within the aquatic part of the system and from organic materials, such as leaves, pollen, and leachates produced in the terrestrial part of the system. In some systems, such as lakes with small, nonforested watersheds, virtually 100 percent of the available energy derives from autochthonous photosynthesis, whereas in other systems, such as small, headwater streams in heavily forested regions, almost all the fixed energy derives from organic detritus of terrestrial origin. But whatever its origin, the fixed energy in organic substances is the driving force that enables the organisms present to metabolize and carry on their life processes. As the energy is used in metabolism it is transformed into heat and dissipated from the system. Hence, unlike nutrients, energy cannot be recycled. It is a one-way street, but like nutrients there must be a continuous supply for the ecosystem to function.

(4) Taking into consideration regional differences in water chemistry and nutrient supply and differences between water bodies in energy availability and efficiency of nutrient recycling, each aquatic system has accumulated over time a diversified biota consisting of many species of organisms ad-

justed to the particular set of conditions in the water body in question. For purposes of analysis and construction of models, these organisms are often clustered into such functional groups as primary producers, herbivores, detritivores, carnivores, decomposers, et cetera, but all are interrelated. That particular species occur in a given lake or river is partly a matter of the species pool of the region and the dispersal capabilities of the individual species, partly a function of the biotic and abiotic relationships in the water body. Although we consider these systems to be in a steady state, intuitively we expect the biota to adjust to long term changes in climate, vegetation, soil development, and internal trends within the system itself, and we also expect the system to be able to accommodate and eventually recover from such short term natural stresses as scouring flushouts in rivers, episodes of volcanism, landslides, and so forth. Homeostasis is restored.

This, to me, is what is meant by the integrity of water—the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a composition and diversity comparable to that of the natural habitats of the region. Such a community can accommodate the repetitive stresses of the changing seasons. It can accept normal variations in input of nutrients and other materials without disruptive consequences. It displays a resistance to change and at the same time a capacity to recover from even quite major disruptions.

My assignment is to consider what history tells us about the response of aquatic systems. Anything that happened in the past is history. Even the words I speak become a part of history as soon as they are spoken. But most of history is unrecorded and hence unavailable for interpretation. In the case of aquatic systems there are anecdotal accounts of particular events or conditions that may have some comparative value. There may be time series of accumulated data for particular rivers or lakes that document what happened during these intervals. And, in the case of lakes (and oceans), the accumulated sediments constitute an historical record of changing climate and watershed conditions and the integrated response of the lake to these changes. Where no previous studies on particular lakes exist and likewise no isolated anecdotes about particular events, the only means we have of interpreting previous conditions is from the sediments. For rivers this possibility does not exist at all, as there is no long term sequential accumulation of sediments. Hence, here we are completely dependent on the written record, except for the geomorphic and hydrologic changes that can be interpreted from the landscape and residual sediments of the valley.

I do not intend to say much about rivers. Their response to point source additions of domestic and industrial wastes is the establishment of a longitudinal gradient involving a succession of chemical processes and organisms, which for organic wastes is sufficiently predictive that a series of zones—the sabrobic system—has been set up to help describe and interpret the process of recovery. Other zone designations have been devised for various kinds of industrial wastes and the responses they elicit.

Organisms vary greatly in their sensitivity to environmental changes accompanying pollution. Fishes together with a majority of insects and molluscs are most sensitive. Blue-green algae and a few miscellaneous animals from several groups are most resistant. These differences in tolerance lead to a greatly simplified community at the point of maximum impact, with the organisms tolerating the conditions here often occurring in tremendous numbers, and then to a gradual buildup in diversity of species and equitability in numbers of individuals downstream. Various diversity indices have been proposed to help quantify these changes. Diatoms are particularly useful in stream studies and their truncated log-normal distributions are useful in assessing the severity of pollution. The experienced investigator can often determine quite easily from the macroinvertebrates present what the stage of recovery is, and can also detect residual effects of pollution, as from lead mines in Wales, that are no longer detectable chemically.

Lakes are fundamentally different from rivers in a number of respects that affect the integrity of water as I have defined it. In the first place, their water movements are not gravity-controlled, unidirectional flows which continually flush out the channel with new water from above, but rather wind-induced circulations. Typically in summer, when the wind is not adequate to overcome the differences in density set up by surface warming, the lake becomes divided into an upper circulating epilimnion and a lower zone, the hypolimnion, cut off from the surface by a steep density gradient and as a result subject to generally much weaker water movements than the epilimnion. During periods of calm weather those lakes that circulate continuously over summer can become temporarily stratified and even the epilimnion of the others can develop secondary stratifications under these circumstances. Regardless of the duration of such stratification, the hypolimnion, or its equivalent in temporary stratification, experiences cumulative chemical changes, most important of which is the gradual depletion of dissolved oxygen by biological activity. The longer the duration of stratification and the greater the amount of biological activity, the more severe will be the oxygen depletion with its attendant stresses on organisms requiring certain levels of dissolved oxygen for their survival.

Unlike rivers, lakes accumulate sediments progressively and sequentially. One effect of these sediments is gradually to reduce the volume of the hypolimnion over time and hence the total volume of dissolved oxygen it contains when stratification becomes established in spring or summer. Consequently, even without any increase in biological activity, the hypolimnion will experience a gradual reduction in oxygen concentration over time, which brings about the extinction and replacement of various deepwater organisms as their tolerances for low oxygen are exceeded.

The sediments constitute a storage for energy and nutrients. Some of this is utilized by bacteria which can continue their activity even to considerable depths in the sediments, or by various animals, which because of their need for molecular oxygen are confined generally to the uppermost few centimeters. Whether the sediments are functioning chiefly as a sink or as a reservoir for nutrients is important in problems concerning eutrophication and its management.

The sediments also constitute a chronological record of processes in the lake and conditions in its watershed, including climate. A perceptive reading of the record—its chemistry, physics, and paleontology—gives us much insight into the stability of lake systems when subjected to various stresses, including those resulting from man's activities, and their rates of recovery.

A third major difference between rivers and lakes is that the water in lakes has a certain residence time, up to 100 years or more in some of the large lakes, determined by the relationship between the input of water from the catchment area and direct precipitation and the total volume of the lake. This allows for the recycling of nutrients in the same place, subject to the constraints imposed by stratification, and the buildup of a diverse community of small floating organisms—the plankton. And even apart from any storage function of the sediments, the residence or replacement time means that there is an inherent lag in response of the system to any increase or decrease in inputs of nutrients or other substances having biological effects. In streams the response to input changes can be almost immediate. Any storages in the sediments are mostly temporary, as the sediments can be swept downstream during the next flood stage.

What I should like to do now is present a few examples of the kinds of responses made by lakes to various stresses.

It was almost axiomatic in limnology until quite recently that lakes increase in productivity over time through natural causes, a process that has been termed natural eutrophication. This idea seemed to be substantiated by some early studies in paleolimnology which showed that the organic content of the sediments increased exponentially over time from a very low level initially to a certain plateau level—the trophic equilibrium—which was then maintained essentially unchanged almost to the present. The trophic equilibrium was regarded as a state in which production was no longer limited by nutrient supply but rather by such factors as light penetration that affect the efficiency of utilization and recycling of nutrients within the system.

The sedimentary chlorophyll degradation products (SCDP) in sediments originate almost entirely from photosynthetic plants, chiefly algae, in the lake itself. Present evidence suggests that these organic compounds are relatively stable in sediments. Hence, the quantitative changes over time of these substances can give an indication of the magnitude and changes in productivity experienced by a lake. One core from Pretty Lake, Ind., (Figure 1), shows low SCDP and hence low productivity in late glacial time and then an exponential increase to a maximum, maintained essentially at plateau level almost to the present. This corresponds to the classical interpretation of the trophic equilibrium in lake ontogeny. But the second core from shallower water shows a decline in SCDP after the maximum following the exponential increase, which does not fit the model.

We now know from this and other studies in paleolimnology that change in productivity over time is not unidirectional from low to high in all lakes, but that some lakes had a period of high productivity initially and then became less productive subsequently. Others had discrete episodes of higher productivity from whatever cause. For example, Lake Trummen in southern Sweden (Digerfeldt, 1972) had high accumulation rates of organic matter, nitrogen, and phosphorus at the beginning of postglacial time approximately 10,000 years ago. These subsequently declined and remained low up to very recent time, when industrial organic effluents completely changed the character of the lake (Figure 2). These relationships are interpreted as reflecting the high early availability of nutrients from the youthful soils of the regional till sheets, with the subsequent decline resulting from the progressive impoverishment of the soil by leaching and by the reduction of subsurface inflow into the lake as basin-sealing sediments accumulated.

Hence, the productive status of a lake is depend-

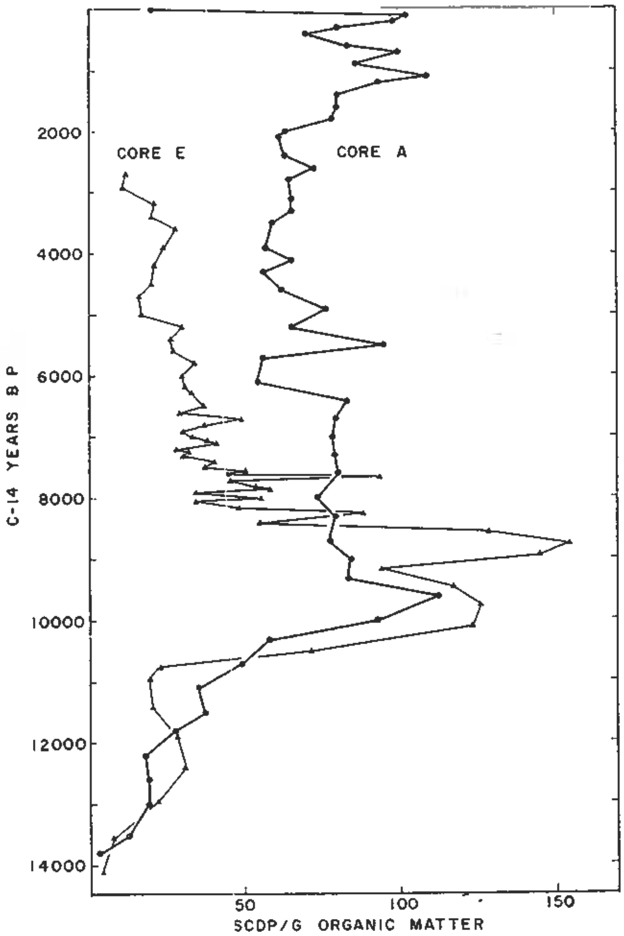


FIGURE 1

ent on the magnitude of its nutrient inputs, subject to the internal controls of the system. If we can decrease the nutrient supply, we can expect a more or less commensurate decrease in productivity. Various attempts are being made to model the magnitude of the response expected from any reduction in nutrient loading, but the rate of response is still unpredictable. The rapid reduction of phosphorus and productivity in Lake Washington following the elimination of secondary sewage effluents (Edmondson, 1972) is encouraging, although some other components of the system, such as nitrogen, did not behave in the same dramatic way. Other examples to be presented suggest that the response time of the total system, or perhaps better the rebound time from a stressed condition, can be much longer than in Lake Washington.

The responses of a lake to the decreasing oxygen concentration of the hypolimnion over time are instructive and significant. Western Lake Erie is so shallow that it stratifies only temporarily in summer during calm weather. Already by 1953 the oxygen demand of the sediments had become such that during a brief period of temporary stratification in late summer the oxygen content of the water overlying the bottom was sufficiently reduced to cause the wholesale death of the nymphs of the burrowing mayfly, one of the most abundant organisms here and a very important fish food (Britt, 1955). The mayflies never reestablished their populations but they have been replaced by smaller oligochaete worms capable of enduring quite low concentrations of dissolved oxygen. Thus, a single event, although obviously with antecedent conditions, led to a complete change in one portion of the biotic community.

The cisco is another case in point, although perhaps less spectacular. If we want to talk about endangered species, or at least endangered populations, this is one. It is a fish that lives in deep water with requirements for both low temperature and high oxygen. If either of these limits is exceeded, the fish perishes. As the summer oxygen concentration of the hypolimnion gradually decreases over time, the cisco, in order to meet its oxygen needs, is forced upward into strata with progressively higher temperatures. Eventually the combination of low oxygen in deep water and high temperatures toward the surface eliminates the habitat suitable for the cisco and the population is extinguished. In 1952 Indiana had 41 lakes with known cisco populations (Frey, 1955a). It is certain that a number of these populations have been completely extirpated since then, and it is not at all certain how long the others will survive.

The species of midge larvae associated with deep-water sediments have different requirements for dissolved oxygen, so that as the oxygen content of the hypolimnion gradually declines over time, the composition of the midge community likewise changes progressively in favor of species capable of tolerating lower oxygen concentrations. This led early in limnology to the establishment of a series of lake types based on the dominant species of offshore midges and presumably representing stages in a successional series. Fortunately the head capsules of the midge larvae, which are well preserved in lake sediments, suffice to identify the organisms to the generic and sometimes to the species levels. In general, the pattern of succession in an individual lake corresponds to the model, with species requiring high levels of oxygen occurring early in the history of the lake; these subsequently are replaced by species more tolerant of reduced oxygen; they in turn are replaced by species still more tolerant, and so on until the only species left is a mosquito-like larva *Chaoborus*, which can endure

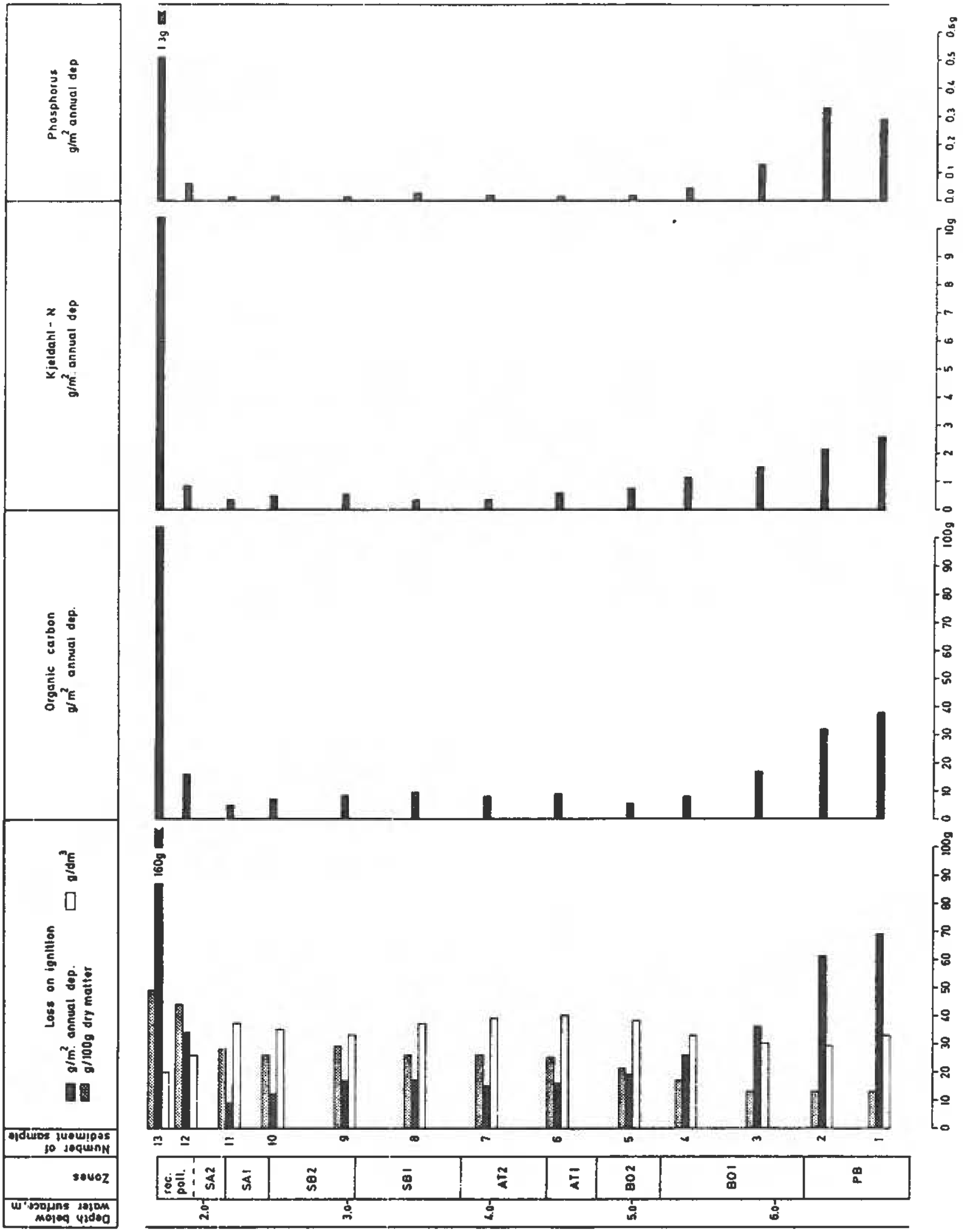

Phosphorus
g/m² annual dep
1.3g
0.0 0.1 0.2 0.3 0.4 0.5 0.6g
Kjeldahl - N
g/m². annual dep
0 1 2 3 4 5 6 7 8 9 10g
Organic carbon
g/m² annual dep.
0 10 20 30 40 50 60 70 80 90 100g
Loss on ignition
g/m². annual dep.
g/100g dry matter
g/dm³
160g
0 10 20 30 40 50 60 70 80 90 100g
Number of sediment sample
13 12 11 10 9 8 7 6 5 4 3 2 1
Zones
rec. poll.
SA2
SA1
SB2
SB1
AT2
AT1
BO2
BO1
PB
Depth below water surface, m
2.0
3.0
4.0
5.0
6.0

FIGURE 2

anaerobiosis for a while, but eventually even it is eliminated if conditions continue to deteriorate.

The most incisive study to date is that of Hofmann (1971) on Schöhsee in northern Germany. A midge community associated with moderate oligotrophy dominated the offshore community until early sub-Boreal time about 1500 B.C. This was followed by a transitional community lasting perhaps 2,500 years, and this in turn by a eutrophic community for the last 1,000 years. The whole story is much more complex than indicated by this too-brief summary in that throughout the 10,000 years of lake history there were migrations of originally shallow-water species into deep water, extinction of deepwater species, and successional dominance of one species or another as conditions gradually changed. Actual quantitative studies of the benthos in 1964-67 compared with similar studies in 1924 show that the populations are still changing (Figure 3). In this interval the population of chironomids, especially *Chironomus*, has declined drastically, being replaced by an increasing population of oligochaetes. *Chaoborus* remained about the same. This situation is reminiscent of western Lake Erie, where oligochaetes took over after the big killoff of mayflies in 1953.

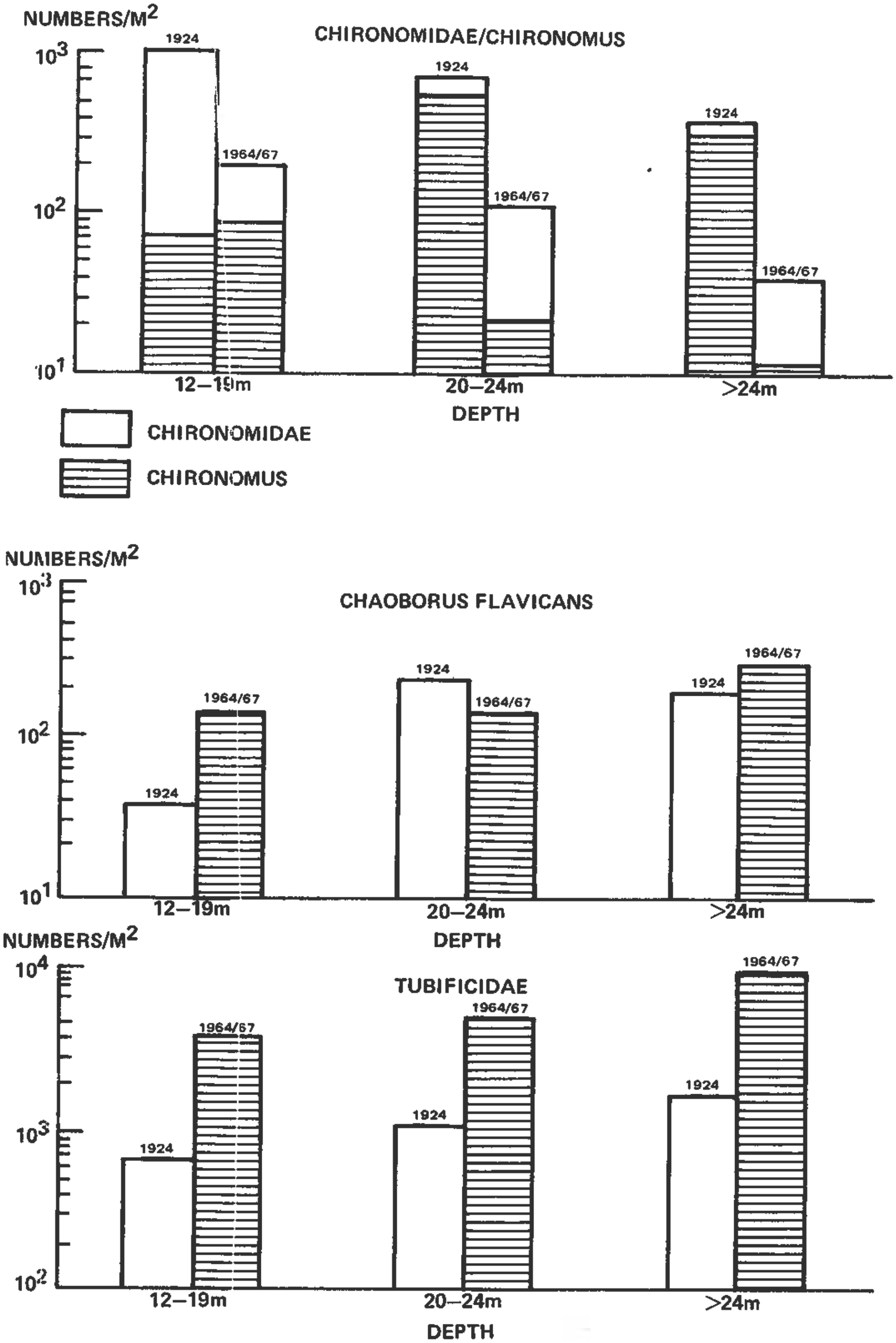
Settlers first moved into the Bay of Quinte region of Lake Ontario about 1784. Government reports describe the devastation of thousands of acres by lumbering and the erosion problems resulting. The initial impact of this land disturbance on the Bay was to change the deepwater sediment from silt dominance to clay dominance and to bring about a marked decrease in organic content through dilution by clay (Warwick, 1975). Subsequently, the organic content increased gradually, although it is still less than pre-impact level, but now there is a pronounced decline in oxygen content of the deep water in summer. The initial response of the midge community was somewhat surprising; it became more oligotrophic than it had been before but then it proceeded through several successional phases to a quite strongly eutrophic stage at present (Figure 4). Unlike previous investigators, Warwick believes that the earliest stages in midge succession are controlled by food supply more than by the minimum annual concentration of oxygen in the hypolimnion. The latter is important chiefly in the later stages of succession. Besides the shift in lithology from silt to clay, the sediments deriving from the impact period are marked by the appearance of the pollen of *Ambrosia* (ragweed), the abundance of which in the sediments roughly parallels but lags somewhat behind the curve showing increase in population of the region. *Ambrosia* provides an excellent time-stratigraphic marker in eastern North America for forest clearance and the initiation of agriculture.

Man's effect on our water resources is nothing recent. Figure 5 is a pollen diagram of Längsee in southern Austria (Frey, 1955b), a lake that presently has a layer of water at the bottom that does not participate in the circulation of the rest of the lake in spring and autumn—a condition known as partial circulation or meromixis. At a particular level in the sediments, which is obvious in the diagram, there are sudden changes in the non-tree pollens, including the appearance of various agricultural weeds and occasional grains of such cultivated plants as cereals and walnut, as well as a disruption in the development of the forest vegetation. At this same level discrete bands of clay occur, separated by a black reduced sediment completely unlike the stable sediment deposited prior to this but identical to what occurs above. Quite obviously, this is when agriculture began in the region about 2,300 years ago, and just as obviously the clearance of the land for agriculture resulted in the inwash of large quantities of clay into the lake, triggering the condition of partial circulation, now maintained by biological means. Hence, the sudden import of large amounts of clay into a lake can have different consequences for different systems.

Lago de Monterosi, a small volcanic lake in central Italy about 40 km from Rome, had an initial small burst of productivity when formed about 25,000 years ago, then a long phase of low productivity up to Roman time, when the construction of a road, the Via Cassia, in 171 B.C. completely changed the input of nutrients and other substances from its small watershed (Hutchinson, et al. 1970). The lake responded by dramatic increases in productivity and sedimentation rates which did not peak until almost 1,000 years after the disturbance (Figure 6). Since then, productivity, as inferred from the accumulation rates of such substances as organic matter, nitrogen, et cetera, has subsided to a level not much greater than that existing before the disturbance. The lag in response and the long duration of the response are probably related to the fact that Monterosi is a closed basin with no permanent streams draining its very small watershed and with output only via seepage.

Grosser Plöner See is a lake in northern Germany famous for the many studies in limnology conducted there by August Thienemann and his associates. In 1256 A.D. a dam constructed at the outlet raised the water level about 2 meters, overflooding much flatland in the process and greatly increasing the extent of the littoral zone. The response of the lake was spectacular (Ohle, 1972). The sedimentation rate, which had increased slowly from about 0.1

SCHÖHSEE BENTHOS 1924 AND 1964/67



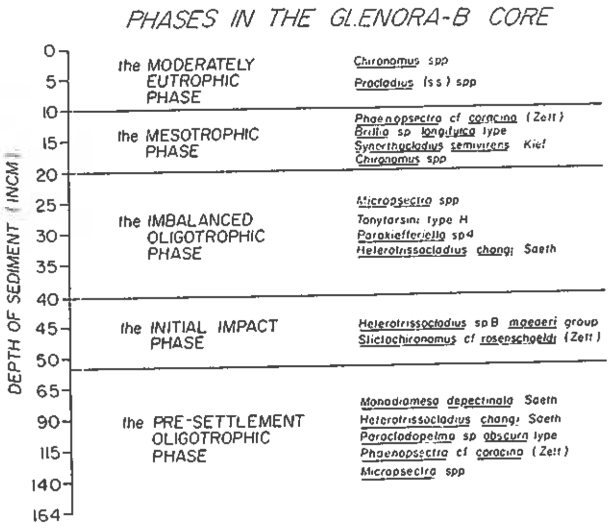


FIGURE 4

mm per year at the beginning of postglacial time to only about 0.5 mm per year 9,000 years later, suddenly jumped 20-fold to more than 10 mm per year. This has resulted in half the 15 meters of sediment in the lake deriving from only the last 700 years of the lake's existence. The big increases at this time in the accumulation rates of such substances as zinc, copper, cobalt, and aluminum reflect the increased input of mineral substances to deep water from the overflooded land and probably from the watershed' also. Correspondingly big increases in the accumulation rates of organic matter, chlorophyll derivatives, and diatom silica reflect the big increase in production within the system resulting from this changed regime (Figure 7).

The lake level had been raised to power a mill dam, as was common in northern Germany at this time, and also to facilitate the production of eels, but the concomitant flooding of valuable agricultural lands resulted in a long-continuing strife be-

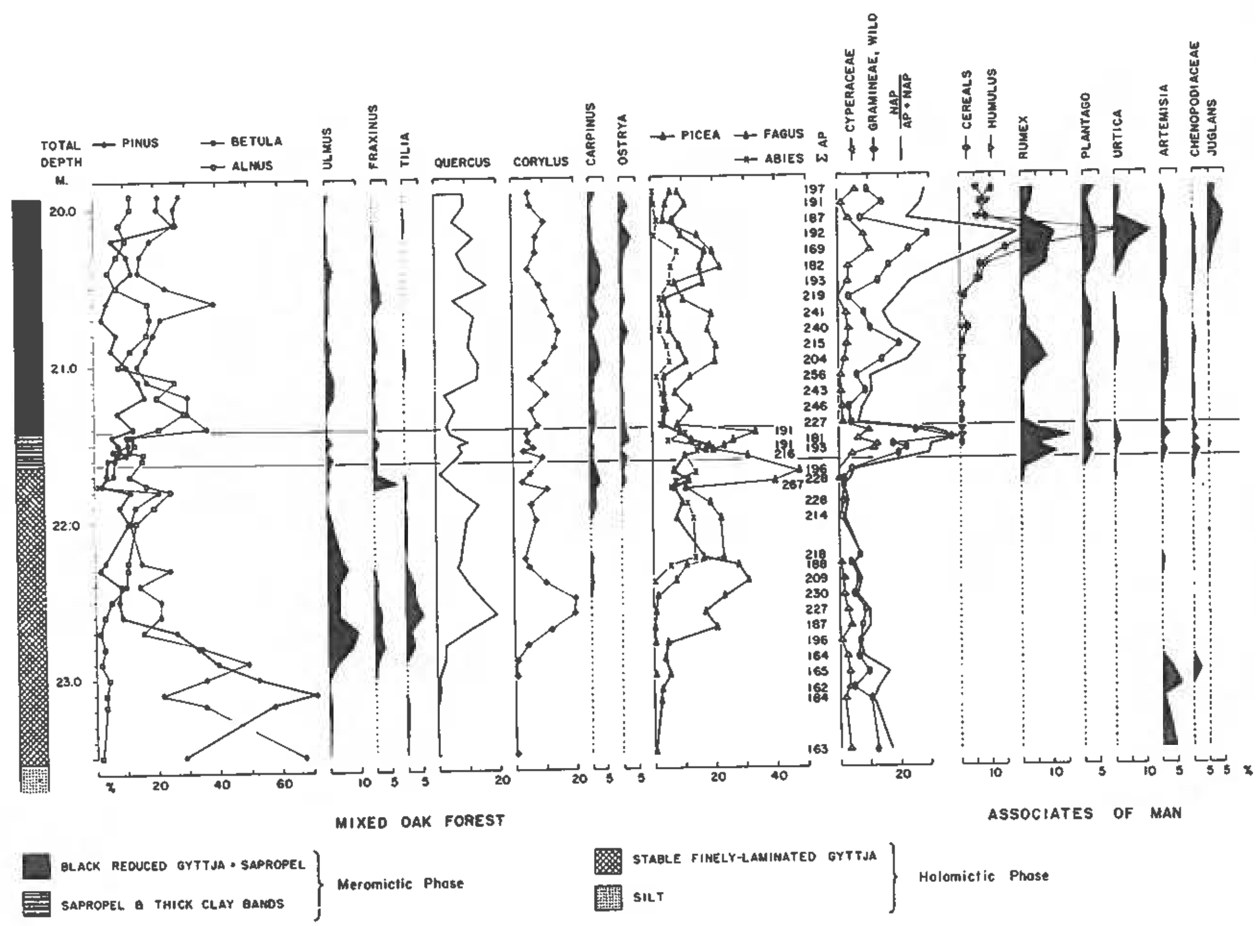


FIGURE 5

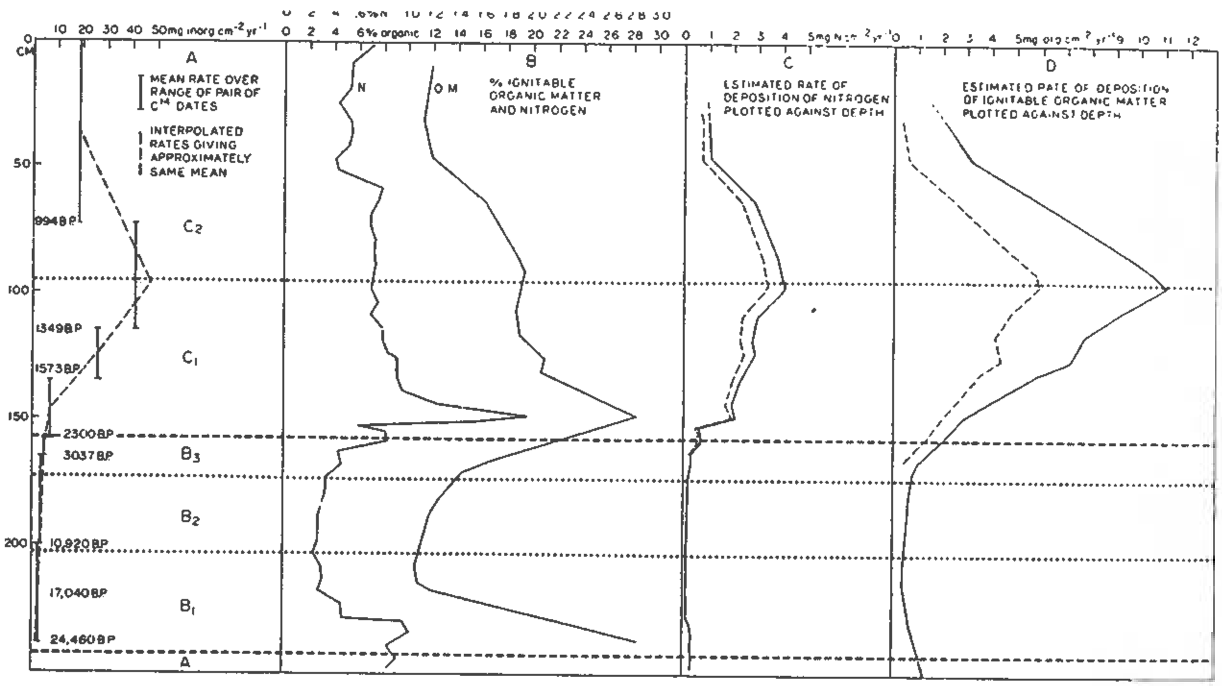


FIGURE 6

tween the mill operators and fishermen on the one hand and the manor owners and farmers on the other. Finally, in 1882 the lake level was lowered by 1.14 m. Up to this time the accumulation rates of most mineral substances had been declining irregularly, and likewise the indicators of biological activity. The sudden lowering in lake level resulted in the erosion and deposition offshore of sediments that had accumulated in shallow water, yielding a discrete horizon of coarse-grained sediments and associated sharp peaks of various mineral constituents. Accumulation rates of chlorophyll derivatives and diatom silica declined at this time, perhaps through light limitation of production by increased inorganic turbidity. The large increase in chlorophyll derivatives in very recent time, reflecting high productivity, is attributed to the heavy use of agricultural fertilizers and phosphate detergents and to the draining of the surrounding wetlands. Such an increase of organic matter and other indicators of production toward the surface is commonplace among lakes being stressed by man, frequently resulting in a completely different type of sediment than anything deposited earlier.

Grosser Plöner See is but one of a number of instances where the productivity of a lake has been markedly increased by raising its water level. The present high productivity of Grosser Plöner See is shared by many lakes of the region, all accomplished within the past few decades in direct response to man's increasing impact on the systems. Ohle (1973) has used the term "rasante Eutrophierung" (racing eutrophication) for this rapid response of lakes to cultural influences, in contrast to the generally slow, balanced development occurring naturally.

The most abundant animal remains in lake sediments are the exoskeletal fragments of the Cladocera, particularly the family Chydoridae (Frey, 1964). They are abundant enough for the construction of close-interval stratigraphies similar to those of pollen and diatoms and for the calculation of various diversity indices and distribution functions. Since the deepwater sediments represent an integration over time and habitat, the population of remains recovered from the sediments is partly artificial, in that all the species represented probably did not co-occur at the same time and place. Yet the diversity indices of the chydorids do show certain demonstrable relationships to such variables as productivity and transparency and, as shown in Figure 8, the relative abundance of the various species in an unstressed situation conforms almost precisely to the MacArthur broken stick model for contiguous but non-overlapping niches (Goulden, 1969a). Hence, the species distribution predicted by this model can be used to assess the extent of imbalance in the system.

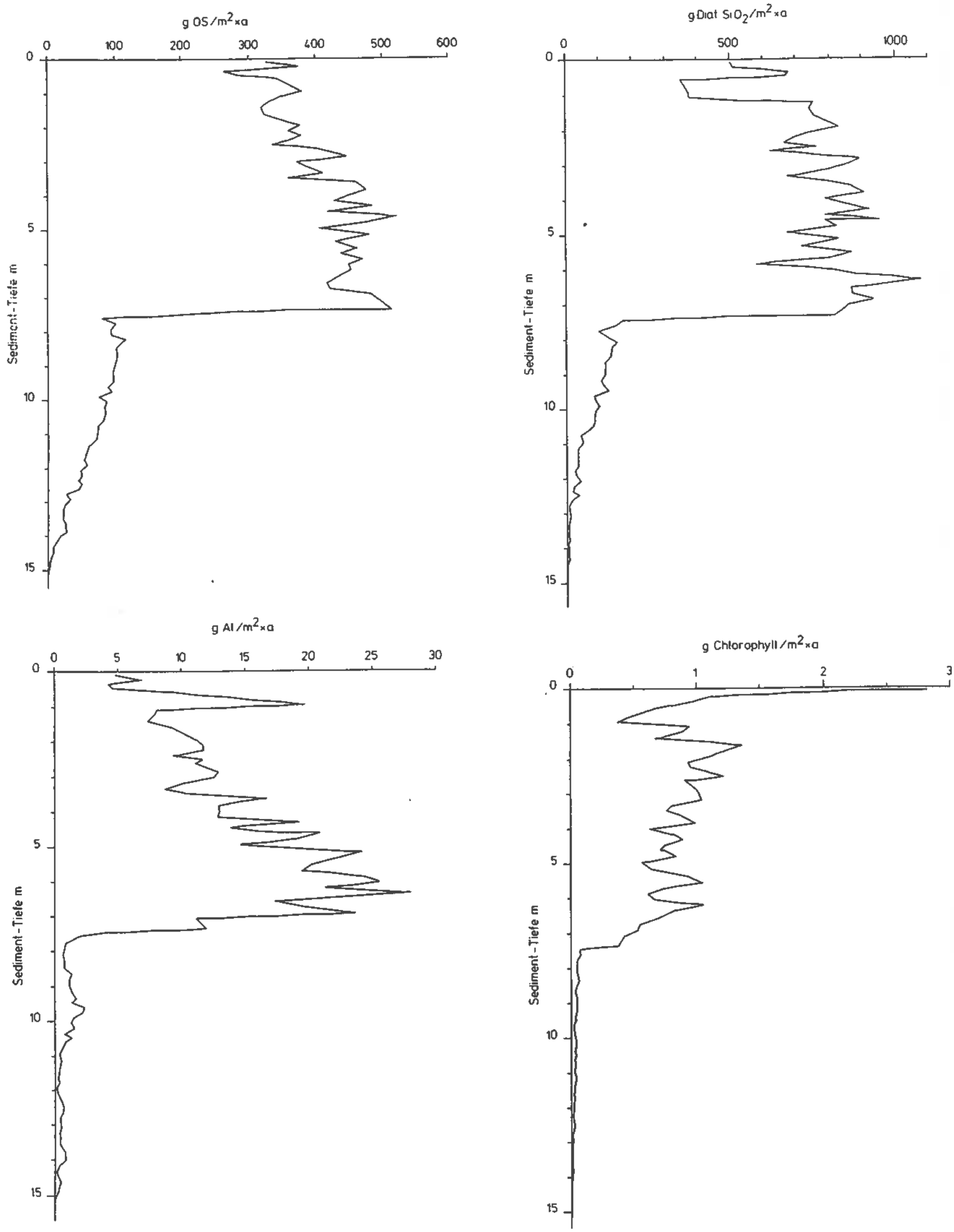

g OS/m²×a
0 100 200 300 400 500 600
Sediment-Tiefe m
0 5 10 15
g Diat SiO2/m²×a
0 500 1000
Sediment-Tiefe m
0 5 10 15
g Al/m²×a
0 5 10 15 20 25 30
Sediment-Tiefe m
0 5 10 15
g Chlorophyll/m²×a
0 1 2 3
Sediment-Tiefe m
0 5 10 15

FIGURE 7

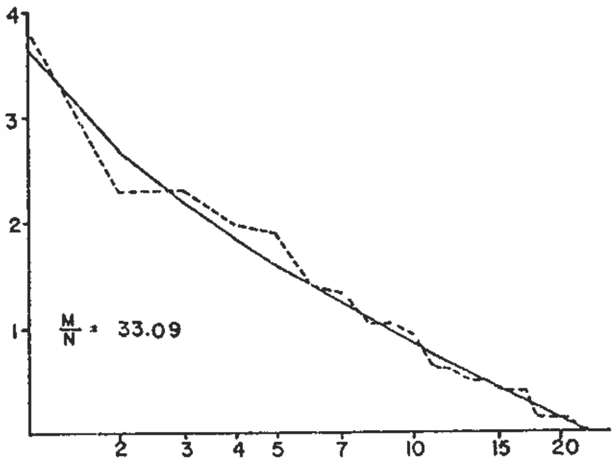


FIGURE 8

In a series of 21 lakes in Denmark for which measurements of annual phytoplankton photosynthesis by radiocarbon uptake are available (Whiteside, 1969), there is a direct relationship between species diversity and transparency and an indirect relationship between species diversity and productivity. There is also an inverse relationship between transparency and productivity. The interpretation of these relationships is that as lakes become more productive, they become less transparent from the development of larger phytoplankton populations, and with higher productivity the chydorid community is thrown out of balance, quite possibly from a reduction of habitat diversity through the curtailment of species diversity and areal extent of the aquatic plants which form the major habitat of the chydorids. And since the chydorids are but one component of a complex community, one can assume that the community as a whole has been stressed by an increase in productivity.

In another study in Denmark, Whiteside (1970) attempted to establish the predictive value of chydorid communities for lake type, and then attempted to use these results in interpreting changes in lake type in postglacial time in response to climate and vegetational patterns of the watersheds. A hard water, eutrophic lake (Esrom Sø) was sufficiently buffered internally that it went placidly about its business during postglacial time almost regardless of external stresses that would be expected to have repercussions on the system, whereas a soft water, oligotrophic lake (Grane Langsø) reacted nervously to even small external stresses. Thus, the response to a given stress can be expected to vary greatly from lake to lake depending on its particular suite of ecological conditions and balances.

The MacArthur predictive model has been used to assess community stresses resulting from the rapid climatic change of the last interstadial (Goulden, 1969a), from episodes of Mayan agriculture in Central America (Goulden, 1966), and from volcanic ash falls in a lake in Japan (Tsukada, 1967). The last study (Figure 9) is interesting in showing that a single instantaneous but massive perturbation, as from an ashfall, can have marked and long-lasting effects on the community structure of a lake.

There are quite a few other studies on the responses of lakes to stresses that might be cited, but I should like to give just one more. The paleolimnology of North Pond in northwestern Massachusetts is being studied intensively by Tom Crisman, a graduate student at Indiana University. Many major changes, almost as precipitous as those in Grosser Plöner See, occurred in the lake shortly after the pine forest represented by pollen zone B was replaced by deciduous hardwoods. Productivity in the lake, as evidenced by the quantity of chlorophyll derivatives in the sediments, increased dramatically at that time, along with nitrogen and phosphorus. A species of planktonic Cladocera, *Bosmina coregoni*, which is usually considered characteristic of more oligotrophic situations, was replaced almost instantaneously by *Bosmina longirostris*, characteristic of more eutrophic situations (Figure 10). Since there is no clear evidence for any major fluctuation in water level and since it is unlikely the Amerindians could have modified the watershed to any appreciable extent, the only correlate and possible cause is the shift in forest composition. But this is difficult to reconcile with the data, because watershed studies to date have demonstrated that deciduous forests are more parsimonious than coniferous forests in releasing nutrients from the system.

Let me attempt to summarize some of the major points developed. Lakes change biologically during their existence from changing inputs of nutrients and energy and from changing internal control mechanisms, associated in part with stratification and depletion of oxygen content in deep water. The biological changes in many instances have been gradual, although in others they have been sudden, associated with natural catastrophes, major changes in water level, or even changes in the dominant vegetation type in the watershed.

Lakes vary in their sensitivity to external stress and in their rapidity and magnitude of response. Man's chief impact is to stress the systems so severely that they are thrown out of balance and the

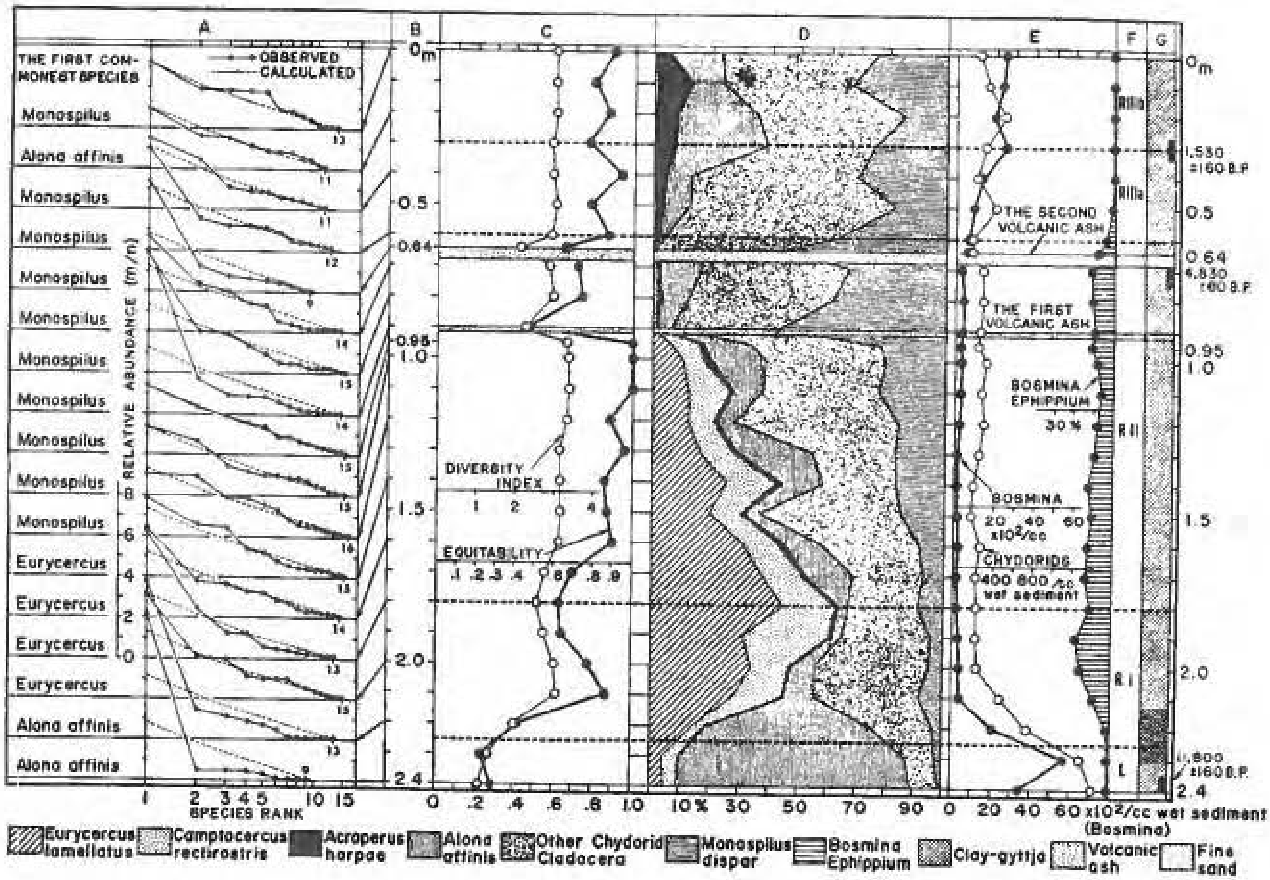


FIGURE 9

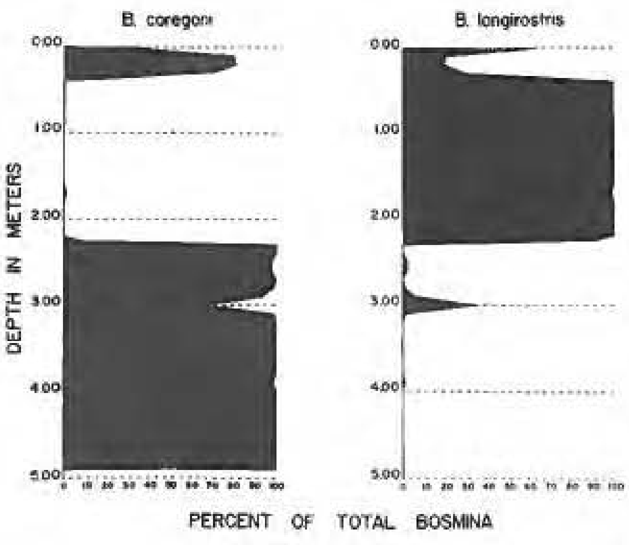


FIGURE 10

rate of change is accelerated—what Ohle calls "rasante Eutrophierung." Both for natural and man-induced stresses, the response of the total system may be fast or slow, and likewise the rate of recovery. The lag may be considerably greater than predicted from the water replacement time, amounting to hundreds of years even in small lakes if our examples from the past have been correctly interpreted. Hopefully, the response time, particularly of recovery, will be fairly short, but faced with the unpredictability of the response time, we should be much more solicitous about the stresses placed on our lakes, as even with massive engineering input they may not recover as rapidly as hoped.

Eutrophication occurs naturally, but so does the contrary process of oligotrophication. That is, a lake can become less productive with time, if its nutrient budget is decreased. Paleolimnology has not yet been able to resolve what the major controls of productivity have been in the past for any particular lake, except by inference from our knowledge of present controls. But since phosphorus, more than any other single substance, is the dominant control of productivity in temperate lakes, it is essential to keep phosphorus inputs at a minimum if we are to have any hope at all of maintaining the integrity of our lakes.

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DISCUSSION

Comment: Your definition of the integrity of water seems to be the capability of maintaining and supporting a composition of organisms that can exist in its natural state. In your discussion you described varying natural states and change of process. How does that translate to a useful definition today?

Dr. Frey: We had an example a couple of weeks ago when two young Ph.D.'s who were modeling ecosystems gave seminars at Indiana University. They had linear models which didn't allow for any change over time. However, in any particular lake there will be changes over time, induced by changes in climate and vegetation, soil development, and so forth. The response of the aquatic community to these changes will probably be adaptive adjustments in species composition and in the relative abundance of species, controlled in part by the mobility of the species. I think a definition of integrity has to include the concept of a balanced, integrated, adaptive community.

Comment: Did you go into these?

Dr. Frey: No, they are objectives.

Comment: Have you made any comparisons with other organisms over an historical period? Would you be able to use changes in the plankton population to detect changes in the water, as you pointed out is possible with fish populations? Would the changes be subtle or quite apparent?

Dr. Frey: I didn't go into any of the long term studies, because even the best of these are less than 100 years old. I know, for example, that there are records for the Chicago water supply which document the kinds and quantities of plankton in southern Lake Michigan over many decades. Probably the longest and most nearly continuous record of all is that for Lake Zürich in Switzerland. Here the deepwater sediments have been accumulating as discrete annual layers since the late 1800's. As the lake became more eutrophic under man's influence, various species of algae invaded the lake and developed to bloom proportions. These are documented by studies of the plankton. Significantly, the blooms of the various species, particularly the diatoms, are also recorded in the appropriate annual layers, so that Lake Zürich constitutes to some extent a calibration system for the interpretation of real events and changes in a lake from what is recoverable from the sediments.

Most of these long term data series have been reported elsewhere at various times. I didn't attempt to summarize them, but instead concentrated on the kinds of interpretations that can be made from the sedimentary record.

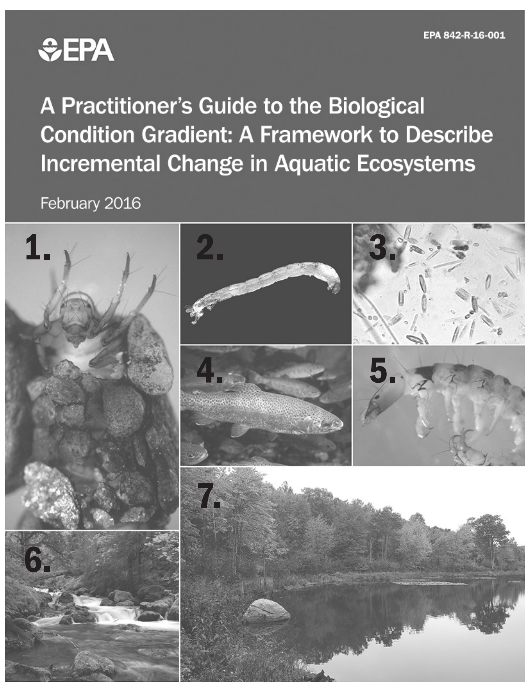
Comment: I'd like to ask you a philosophical question stemming from your definition of integrity. In your opinion, are efforts to reverse a naturally occurring trend toward eutrophication counter to the integrity of that lake?

Dr. Frey: I had to leave out a number of pages of my prepared text because of time limitations (but these are included in the published paper). For the chydorid cladocerans, which are well represented in lake sediments, the species diversity of the community declines as the productivity of the lake increases, indicating that the system is being stressed. This should not be interpreted to mean that all productive lakes are out of balance because the rate of change is probably the important consideration. Where the increased productivity is the result of man or of some essentially instantaneous event such as a volcanic ash fall, the rate of change in nutrient budgets or other environmental conditions is so great that the community cannot keep pace with orderly and adaptive adjustments. But where the forcing variables change slowly over time the aquatic biota is able to maintain an internal balance. Hence, I am in favor of either reversing the trend toward increasing productivity in our natural waters, except where this is specifically desired, or at least sufficiently reducing the rate at which eutrophication is occurring so that the system is not stressed unduly.

EPA 842-R-16-001

A Practitioner's Guide to the Biological Condition Gradient: A Framework to Describe Incremental Change in Aquatic Ecosystems

February 2016



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A Practitioner's Guide to the Biological Condition Gradient: A Framework to Describe Incremental Change in Aquatic Ecosystems

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B1. *Upper Mississippi River: Development of a Biological Condition Gradient for Fish Assemblages of the Upper Mississippi River and a "Synthetic" Historical Fish Community*
Ed Rankin, Midwest Biodiversity Institute, Columbus, Ohio

B2. *Narragansett Bay: Development of a Biological Condition Gradient for Estuarine Habitat Quality*
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B3. *Caribbean Coral Reefs: Benchmarking a Biological Condition Gradient for Puerto Rico Coral Reefs*
Patricia Bradley[**] and Deborah Santavy, PhD, USEPA Office of Research and Development, National Health and Environmental Effects Research Laboratory–Gulf Ecology Division, Gulf Breeze, Florida

B4. *New England: Using the Biological Condition Gradient and Fish IBI to Assess Fish Assemblage Condition in Large Rivers*
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Executive Summary

The Clean Water Act (CWA) established a long-term objective to restore and protect the biological integrity of the nation's waters. In the more than 40 years since the passage of the CWA, there has been considerable progress in the science of aquatic ecology and in the development of biological monitoring and assessment techniques to support implementation of the Act. The U.S. Environmental Protection Agency (EPA) published its first guidance document on biological assessments and criteria in 1990. Since then, aquatic science and its application in state water quality programs has advanced significantly. States, territories, and authorized tribes (herein identified as "states") now routinely use biological information to directly assess the biological condition of their aquatic resources, track changes in their condition, and develop biological criteria to set expectations for maintaining biological integrity.

This document is designed for scientists engaged in biological assessments of water bodies. It outlines a conceptual framework, the Biological Condition Gradient (BCG), for states to use to more precisely define and interpret baseline biological conditions, help evaluate potential for improvement in degraded waters, and measure and document incremental changes in condition along a gradient of anthropogenic stress. The conceptual framework can be populated with state or regional data to develop a quantitative model and establish numeric thresholds. The BCG is intended to complement existing biological assessment and criteria methods and approaches.

What is the Biological Condition Gradient?

The BCG is a conceptual, scientific framework for interpreting biological response to increasing effects of stressors on aquatic ecosystems. The framework was developed based on common patterns of biological response to stressors observed empirically by aquatic biologists and ecologists from different geographic areas of the United States. Scientists from 21 states, one interstate basin association, and one tribe were involved in BCG development, in addition to scientists from EPA, the U.S. Geological Survey, universities, and the private sector. The framework describes how 10 characteristics of aquatic ecosystems change in response to the increasing levels of stressors, from an "as naturally occurs" condition (e.g., undisturbed/minimally disturbed condition) to severely altered conditions. The characteristics, defined in this document as "attributes," include aspects of community structure, organism condition, ecosystem function, and connectivity. The BCG framework can be considered analogous to a field-based dose-response curve where the dose (x-axis) represents increasing level of anthropogenic stress, and the response (y-axis) represents biological condition.

Who Will Use the Biological Condition Gradient and For What Purpose?

Currently most states are using biological assessment information to support their water quality management programs. The BCG contributes to the EPA biological assessment and criteria "toolbox," which includes biological indices, models, statistical approaches, and guidance. The BCG builds upon and complements these approaches to provide a more refined and detailed measure of biological condition and can help water quality management programs to:

- More precisely define and measure biological condition for specific waters;
- Identify and protect high quality waters;
- Evaluate potential for improvement in degraded waters;
- Track changes in condition;

- Develop biological criteria; and
- Clearly communicate the likely impact of water quality management decisions to stakeholders.

These applications support CWA programs such as 305(b) assessments and reports, 303(d) listing of impaired waters, and the Total Maximum Daily Load program implementation. The document includes examples of how states are using, or are considering using, the BCG to support their water quality management programs.

Why Now?

As the first BCG projects have been completed, there has been increasing interest in the BCG by other state water quality management programs. Based on informal discussion with state water quality managers and scientists who have been directly engaged in BCG development, their primary motivation for using a BCG has been to more precisely define baseline conditions, better understand the quality of their reference sites, identify high quality waters as candidates for additional protection, help evaluate the potential for restoration of degraded waters, and document incremental improvements as best management practices are implemented. In all cases, the states have emphasized the value of the BCG to help communicate to the public the biological condition of their waters in context of the CWA integrity objectives and the likely outcomes of water quality management decisions.

Because of the interest in BCGs, it is important now to document the status of model development, discuss current strengths and limitations, and provide examples of how states are developing and applying the BCG. This document provides a template and step-by-step process for constructing robust BCGs, drawing from the lessons learned during a decade of testing by interstate, state, territorial, and local government water quality management programs. As BCG development and calibration continues, it is expected that the BCG process will be refined and improved.

Biological Condition Gradient Development: Decision Rules

This document describes the steps that entail convening an expert panel in order to construct narrative descriptions and quantitative rules for assigning sites to BCG levels. Different approaches to developing quantitative rules are discussed (e.g., mathematical set theory, derivation and calibration of biological indices, and multivariate statistical and/or predictive modeling approaches). The core objective of the panel process is to elicit expert judgment on defining ecologically significant change in the biotic community <u>and to document the underlying rationale for the judgments</u>. By using a process to elicit expert judgment, first narrative and then quantitative rules emerge and are tested and refined based on the current state of the science, expert knowledge, and available data. The intended end product is a set of well-vetted and transparent decision rules that can be readily understood and implemented by state water quality program managers and scientists. Routine use of a quantitative BCG model by state water quality management programs requires well documented and transparent decision rules so that assessments can be made for newly sampled water bodies without reconvening the expert panel.

Specifically, the document presents:

- An approach to quantify the conceptual BCG framework and develop a numeric model. This approach is based on elicitation of the experts' decision criteria and incorporation those of criteria into a numeric decision model using a mathematical set theory approach (e.g., fuzzy logic). This approach has been tested and refined in most of the BCG projects to date.

- Considerations and approaches for relating the BCG with the state's existing biological assessment methods and tools such as multimetric biological indices. To date, most states have developed biological indices.
- An example of a state approach to quantify the conceptual BCG. This approach involves development of statistical models that predict (or simulate) the expert decisions and may or may not use elicited expert reasoning or rules.

Building on these initial efforts, it is expected that additional methods to quantify the conceptual BCG will be identified and tested.

The Stress Axis

The x-axis of the BCG framework, the Generalized Stress Axis (GSA), conceptually describes the range of anthropogenic stress that may adversely affect aquatic biota in a particular area. It is a theoretical construct. As multiple stressors are usually present in a system, the GSA seeks to represent the cumulative stress that may influence biological condition. Typically, states have defined a stress gradient using single or a combination of known, measurable stress gradients that in reality represent a portion of the stressors impacting a water body. The conceptual GSA provides a framework to assist in development of as comprehensive and robust a quantitative stress gradient as possible to support BCG development. A well-defined, quantitative GSA, and the underlying data used to develop it, may serve as a nexus between biological and causal assessments, thereby linking management goals and selection of management actions for protection or restoration. However, a systematic testing of technical approaches to define and apply a GSA to BCG development has not been conducted. This document discusses technical issues to consider and provides examples of approaches to quantify a GSA. Opportunities in the future may include piloting methods for application of national, regional, or basin scale databases and methods to support state efforts to quantify a GSA for a specific geographic region and water body type.

Document Organization

Chapters 1 and 2 explain the purpose and scientific underpinnings of the BCG. Chapters 3 and 4 present methods on how to define and quantify the BCG biological axis, the biological levels of condition that span undisturbed to severely altered conditions. Chapter 5, supported by Appendix A, provides an overview, framework, and examples to describe the stress axis of the BCG model, the GSA. Examples of how states have developed and applied the BCG are presented in Chapter 6. To date, use of the BCG to support water quality management has primarily been for fresh water, perennial streams. However, work underway is presented in Appendix B on BCG development for large rivers, estuaries, and coral reefs.

Chapter 1. Introduction to the Biological Condition Gradient

1.1 Document Purpose

The Clean Water Act (CWA) established a long-term objective to, among other things, restore and protect the biological integrity of the nation's waters (Figure 1). In the more than 40 years since the passage of the CWA, there has been considerable progress in the science of aquatic ecology and in the development of biological monitoring and assessment techniques to support implementation of the Act (USEPA 2011a, 2013a). Since the U.S. Environmental Protection Agency (EPA) published its first guidance document on biological assessments and criteria, aquatic science and its application in state water quality programs has advanced (USEPA 1990, 2002, 2011a, 2013a). States, territories, and authorized tribes (herein referred to as "states") now routinely use biological information to directly assess the condition of their aquatic resources, track changes in biological condition, and develop biological criteria to set expectations for maintaining biological integrity.

Figure 1. Stream and wadeable river.

Under the CWA, states have the primary authority to implement their water quality programs with EPA review for consistency with the CWA requirements, which include implementing regulations. As a consequence, states have independently developed technical approaches to assess biological condition and establish thresholds (Hawkins 2006; USEPA 2002). Although these different approaches have fostered innovation, they have complicated a nationally consistent approach to interpreting the condition of aquatic resources. A consistent approach to interpreting biological condition will allow scientists, water resource managers, and stakeholders to share a common understanding and language to describe the condition of their waters, as well as share data and information across jurisdictional boundaries (Davies and Jackson 2006).

In addition to using a variety of approaches for assessing and interpreting biological condition, states have created a range of different aquatic life use (ALU) classes to describe the expected biological condition of their waters. At one end of the spectrum, states have adopted a general narrative statement that replicates the ALU goal identified in the CWA (e.g., protection and propagation of fish, shellfish, and wildlife). At the other end are more detailed approaches that describe the expected species, assemblages, or habitats (e.g., salmonids, warmwater habitat, coldwater fisheries) or that specify levels of condition (e.g., excellent, good, fair). Currently, most states have established one general ALU class, with a single threshold for assessing attainment. A limitation of a single ALU class is that the full range of biological conditions along a human disturbance gradient is limited to only two categories: pass and fail. Water bodies assigned to a single ALU class could include a range of biological conditions found in undisturbed to moderately disturbed landscapes, or, in some cases even include highly disturbed conditions where anthropogenic impacts are widespread and pervasive. As a result, a water body supporting biological conditions characteristic of higher quality waters could degrade to a lower level of water quality yet still be categorized as meeting its ALU. In contrast, for water that is severely degraded, the designated ALU might not be achievable in the short term, and therefore incremental improvements due to management actions will not be measured or acknowledged. A scientific framework that describes incremental biological changes along the full gradient of human disturbance helps water quality managers identify and protect high quality waters and track incremental improvements in degraded waters.

This document outlines a conceptual framework, the Biological Condition Gradient (BCG), that states can use to more precisely describe existing, or baseline, biological condition; help evaluate potential for improvement in condition; and measure incremental changes in condition along a gradient of human disturbance, i.e., anthropogenic stress. The conceptual framework can be populated with state or regional data to develop a quantitative model. It is intended to complement existing biological assessment and criteria methods and approaches.

This document reports on the current status of quantitative model development and application. As BCG development and calibration continues, it is expected that the BCG process will be further refined and improved.

1.2 Background: When and Why?

In 2000, EPA convened a technical expert workgroup to identify scientifically sound and practical approaches that would help states use biological assessments to better determine existing conditions and potential for improvement, more precisely define ALUs, and develop biological criteria. The workgroup consisted of scientists from federal, state, and tribal water programs, an interstate basin association, the academic research community, and the private sector (see Davies and Jackson 2006 for a list of workgroup members). The overarching objective of this effort was to develop a common framework and language for interpreting biological condition. In the subsequent four years, the workgroup met annually with drafts of the framework undergoing review and preliminary testing between meetings. The effort was primarily guided by the practical experience of scientists and water quality program managers from the 21 states, the interstate basin association and tribe participating in the workgroup. The workgroup developed the conceptual BCG framework to describe levels, or tiers, of biological response to increasing levels of stressors. The conceptual BCG was developed and tested through a series of data exercises using a diverse array of data sets with initial focus on freshwater perennial streams and wadeable rivers.

The workgroup activities coincided with a National Research Council (NRC) review of EPA's Total Maximum Daily Load (TMDL) program and publication of its report *Assessing the TMDL Approach to Water Quality Management* (NRC 2001). Among other recommendations, the NRC recommended the use of biological assessments to better understand water quality and the establishment of a more precise, descriptive approach to goal-setting as a step towards improving decision making and establishing appropriate ALU goals. For example, rather than stating that a water body needs to be "fishable," the ALU would ideally describe the expected fish assemblage or population (e.g., salmonid, coldwater fishery, warmwater fishery), as well as the other biological assemblages necessary to support that fish population. Additionally, levels of expected condition would be defined based on potential of a water body to achieve a higher level of condition (e.g., salmonid spawning versus migration; undisturbed and minimally disturbed conditions versus moderately or highly disturbed). The NRC recommendation to more precisely define designated ALUs was taken into account by the BCG workgroup as they developed the BCG framework. Since completion of the conceptual BCG framework (Davies and Jackson 2006), many states have further developed and refined quantitative BCG models (see Table 4, Chapter 3). In conjunction with other water quality management technical tools, the state programs that have developed and applied the BCG have done so to help:

- Set scientifically defensible, ecologically-based aquatic life goals based on existing conditions and potential for improvement;
- Determine baseline conditions and measure impacts of multiple stressors or system altering conditions (e.g., climate change) on aquatic life;
- Further the use of monitoring data for the assessment of water quality standards (WQS) and tracking changes in biological condition;
- Identify high quality waters for protection (e.g., Tier III antidegradation); and
- Communicate to stakeholders the likely impact of decisions on protection and management of aquatic resources.

When asked about the most immediate, value-added benefits to their water quality management programs from the development of a quantitative BCG model, state water quality program managers and scientists cited the ability to measure and document incremental improvements due to management actions and better identify and protect high quality waters.

The BCG conceptual framework, quantitative model development, and implementation reflects an improved understanding of aquatic ecosystems and their biota resulting from more than 40 years of assessment data and advances in use of these data in state water quality management programs. This document represents the culmination of four years of workgroup deliberations, including four workgroup meetings and two workshops to "road test" the conceptual BCG framework, followed by ten years of development and application of quantitative BCG models in state programs. Over the past ten years, the BCG has been developed for perennial streams, including headwater streams, using expert consensus to develop narrative and numeric decision rules to assign sites to BCG levels. The use of the BCG to complement or refine existing state measures such as Indices of Biotic Integrity (IBIs) is being explored. Application of the BCG to water bodies other than perennial streams is underway for large rivers, estuaries, and coral reefs. These latter efforts show promise for expanding the application of the BCG beyond streams to more complex systems.

1.3 The Biological Condition Gradient: Brief Overview

The conceptual BCG is a scientific framework for interpreting biological response to increasing effects of stressors on aquatic ecosystems (Figure 2). The framework was developed based on common patterns of biological response to stressors observed empirically by aquatic biologists and ecologists from different geographic areas of the United States (Davies and Jackson 2006). It describes how characteristics of aquatic ecosystems that are typically measured by state water quality management programs change in response to increasing levels of stress (see Table 1). The characteristics, defined as attributes, include properties of the communities (e.g., tolerance, rarity, native-ness) and organisms (e.g., condition, function) and are more fully described in Chapter 2.

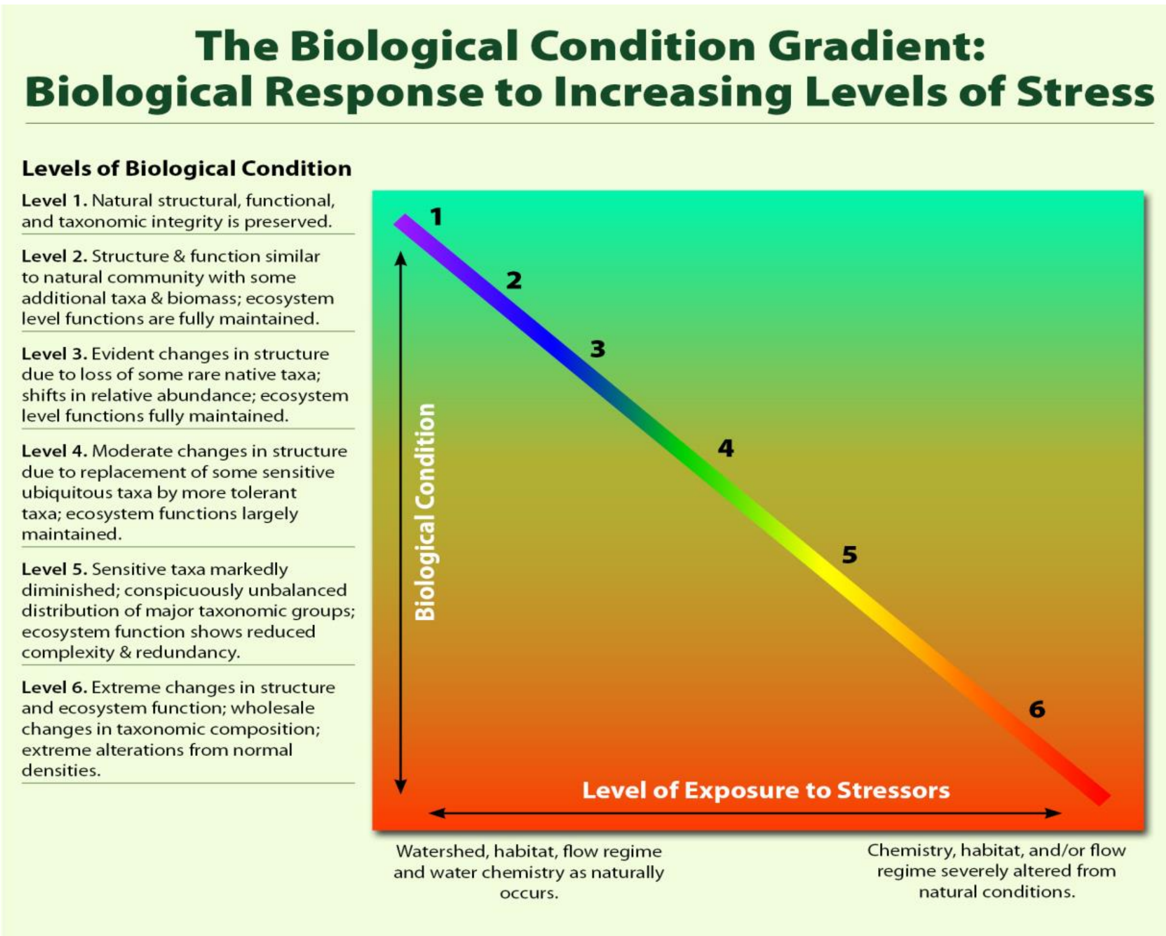

Figure 2. Conceptual model of the BCG. Although in reality the relationship between stressors and their cumulative effects on the biota is likely nonlinear, the relationship is presented as such to illustrate the concept.

The BCG can be considered analogous to a field-based dose-response curve where the dose (x-axis) represents increasing levels of stressors, and the response (y-axis) represents biological condition. Stressors are physical, chemical, or biological factors that induce an adverse response from aquatic biota (USEPA 2000b). For example, high concentrations of certain metals, nutrients, or sediment can adversely impact, or stress, aquatic biota. Loss of suitable aquatic habitat or presence of aquatic invasive species can also adversely impact the aquatic biota expected for a specific water body. These stressors can cause aquatic ecosystems to change from natural conditions and exhibit altered compositional, structural, and functional characteristics. The degree to which stressors affect the biota depends on the magnitude, frequency, and duration of the exposure of the biota to the stressors. Developing a BCG for a given system characterizes the general relationship between its stressors in total and a water body's overall biological condition. Multiple stressors are usually present, and thus, the stress x-axis of the BCG seeks to represent their cumulative influence as a Generalized Stress Axis (GSA),[1] much as the y-axis generalizes biological condition. The x and y axes of the BCG serve as a framework to organize, relate, and help reconcile the mosaic of factors and interactions that exist, parts of which will be characterized and measured using biological, chemical, physical, and/or land use/land cover indicators.

Table 1. Ecological characteristics (i.e., attributes) used to develop the BCG

Attribute	Description
I	Historically documented, sensitive, long-lived, or regionally endemic taxa
II	Highly sensitive taxa*
III	Intermediate sensitive taxa
IV	Intermediate tolerant taxa
V	Tolerant taxa
VI	Non-native or intentionally introduced species
VII	Organism condition
VIII	Ecosystem function
IX	Spatial and temporal extent of detrimental effects
X	Ecosystem connectance

*Note: Identified as *Sensitive-rare taxa* in Davies and Jackson 2006.

The BCG differs from the standard dose-response curve in that the BCG does not represent the laboratory response of a single species to a specified dose of a known chemical, but rather the *in-situ* response of the resident biotic community to the sum of stressors to which that community is exposed. Thus, it is an outcome-based measure and something that can express complex water quality goals such as biological integrity. In this document EPA proposes a BCG that is divided into six levels of biological condition along a generalized stressor-response curve, ranging from observable biological conditions found at no or low levels of stressors (level 1) to those found at high levels of stressors (level 6). States may propose to consolidate or aggregate these levels into fewer levels or further refine and increase the number of levels. Regardless of how many levels a quantitative BCG may ultimately include, it can be crosswalked with the conceptual model. Chapter 6 and Appendix B illustrate examples of ecoregional or state-specific BCGs and how they may be "mapped" onto the conceptual BCG.

Between 2000 and 2005, the original framework was tested at annual workgroup meetings and then at two regional workshops in the Great Plains and in the Arid Southwest. It was tested by determining how consistently the scientists assigned samples of benthic macroinvertebrates or fish to the different levels

[1] For more information on the Generalized Stress Axis, see Chapter 5.

of biological condition in freshwater streams. Workgroup members identified similar sequences of biological response to increasing levels of stressors regardless of geographic area and predicted that the framework in principal should be applicable to other water body types. These results support the development and application of the BCG as a nationally applicable framework for interpreting the biological condition of aquatic systems (Davies and Jackson 2006).

Understanding the links between stressors (and their sources) with the response of the aquatic biota will help water quality managers to more accurately determine both the existing and potential conditions of the aquatic biota in a specific water body and help predict the stressors that affect that condition (Figure 3). This information will assist water quality program managers in determining the most effective recourse to address biological impairment. There are different approaches and new studies, methods, and large data sets that can assist states to better define and quantify the causal sequence between stressors and their sources and biological responses once biological impairment is identified.[2] Ultimately, the goal of the EPA biological criteria program is to build a stronger technical bridge between biological condition assessments, causal assessments, and the actions taken to protect and restore biological condition. A well-defined BCG x-axis, the GSA, and the science underlying it may help achieve this objective. In Chapter 5, information on approaches and technical challenges to define the GSA are discussed, with examples of a conceptual GSA framework and potential stress indicators included in Appendix A.

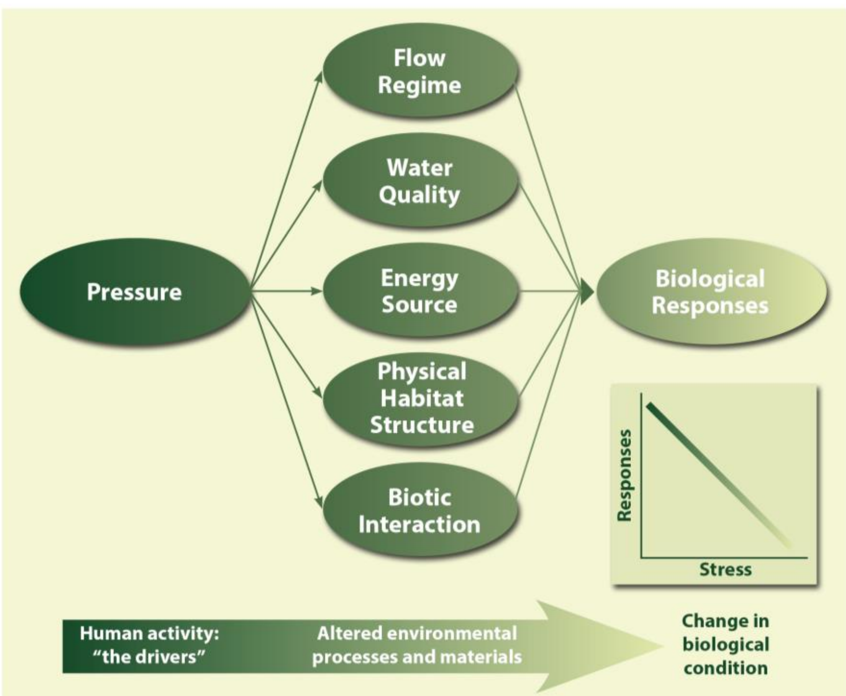

Figure 3. Model illustrating the multiple pathways through which human activities may exert pressure on an aquatic system by altering fundamental environmental processes and materials, creating stressors that may adversely affect the aquatic biota (Source: Modified from figure courtesy of David Allen, University of Michigan).

[2] See http://www3.epa.gov/caddis/ and http://water.epa.gov/lawsregs/lawsguidance/cwa/tmdl/recovery/overview.cfm. Accessed February 2016.

1.4 Use of the Biological Condition Gradient to Support Water Quality Standards and Condition Assessments

The full objective of section 101(a) of the CWA is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters. In the scientific literature, an aquatic system with chemical, physical, and biological integrity has been described as being capable of "supporting and maintaining a balanced, integrated, adaptive community of organisms having a composition and diversity comparable to that of the natural habitats of the region" (Frey 1977).

Over the intervening years, the understanding of how to define and measure the integrity of aquatic systems has advanced considerably. The term "integrity" has been further refined in the literature to mean a balanced, integrated, adaptive system having a full range of ecosystem elements (e.g., genera, species, assemblages) and processes (e.g., mutation, demographics, biotic interactions, nutrient and energy dynamics, metapopulation dynamics) expected in areas with no or minimal human disturbance (Karr 2000). The aquatic biota residing in a water body are the result of complex and interrelated chemical, physical, and biological processes that act over time and on multiple scales (e.g., instream, riparian, landscape) (Karr et al. 1986; Yoder 1995). By directly measuring the condition of the aquatic biota, one is able to more accurately define the aquatic community that is the outcome of all these factors.

To help achieve the integrity objective, the CWA also established an interim goal for the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water. EPA has interpreted the "protection and propagation" interim goal for aquatic life to include the protection of the full complement of aquatic organisms residing in or migrating through a water body. As explained in EPA's *Water Quality Standards Handbook* (USEPA 2014a), the protection afforded by WQS includes the representative aquatic community (e.g., fish, benthic macroinvertebrates, and periphyton):

> The fact that sport or commercial fish are not present does not mean that the water may not be supporting an aquatic life protection function. An existing aquatic community composed entirely of invertebrates and plants, such as may be found in a pristine tributary alpine stream, should be protected whether or not such a stream supports a fishery. Even though the shorthand expression 'fishable/swimmable' is often used, the actual objective of the Act is to restore the chemical, physical and biological integrity of our Nation's waters (section 101(a)). The term 'aquatic life' would more accurately reflect the protection of the aquatic community that was intended in section 101(a)(2) of the Act.

The representative community of aquatic organisms residing in, or migrating through, a water body will vary depending on the water body type. For example, fish, benthic macroinvertebrates, and periphyton are aquatic assemblages measured by states and tribes when assessing the biological condition of most streams and rivers. However, in headwater streams and many wetlands, amphibians are an important component of the biotic community, and fish may be absent. Large river and estuarine assessments typically include both benthic invertebrates and fish community measures. In coral reefs, coral, sponge, and fish communities are key assemblages to measure and assess. The BCG offers a framework to provide more detailed and descriptive statements of the aquatic community expected in an undisturbed or minimally disturbed aquatic community, as well as potential incremental changes that might be expected in community characteristics with increasing levels of anthropogenic stress.

1.4.1 Use of the Biological Condition Gradient to Support Aquatic Life Use Assessments

While section 101(a) of the CWA establishes the objective to restore and maintain the chemical, physical, and biological integrity of the nation's waters, other sections of the CWA establish the programs and authorities for implementation of this objective. Section 303(c) provides the basis of the WQS program. WQS are components of state (or, in certain instances, federal) law that define the water quality goals of a water body, or parts of a water body, by designating the use or uses of the water body and by setting criteria necessary to protect the uses (in addition to antidegradation requirements).

Although the CWA gives EPA an important role in determining appropriate minimum levels of protection and providing national oversight, it also gives considerable flexibility and discretion to state water quality managers to design their own programs and establish levels of protection above the national minimums. CWA section 303 directs states to adopt WQS to protect the public health and welfare, enhance the quality of water, and serve the purposes of the CWA. "Serve the purposes of the Act" (as defined in sections 101(a), 101(a)(2), and 303(c) of the CWA) means that WQS should include provisions for restoring and maintaining chemical, physical, and biological integrity of state waters; provide, wherever attainable, water quality for the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water (i.e., "fishable/swimmable"); and consider the use and value of state and tribal waters for public water supplies, propagation of fish and wildlife, recreation, agricultural and industrial purposes, and navigation. Further requirements for WQS can be found at 40 *Code of Federal Regulations* (CFR) Part 131.

State WQS provide the foundation for water quality-based pollution control programs. With the public participating in their adoption (see 40 CFR 131.20), such standards serve the dual purposes of (1) establishing the water quality goals for a specific water body and (2) providing the regulatory basis for the establishment of water quality-based treatment controls and strategies beyond the technology-based levels of treatment required by sections 301(b) and 306 of the CWA. The WQS serve as, among other things, the basis for ALU attainment decisions, National Pollutant Discharge Elimination System (NPDES) permit limits, and the targets for TMDLs.[3]

40 CFR Part 131.10(a) of the WQS regulation requires that states specify appropriate water uses to be achieved and protected. A water body's designated uses are those uses specified in WQS, whether or not they are being attained (40 CFR 131.3(f)). *The designated use of a water body is the most fundamental articulation of the water body's role in the aquatic environment as defined by society*. All of the water quality protections established by the CWA follow from the water body's designated use. As designated uses are critical in determining the water quality criteria that apply to a given water body, determining and clearly defining the appropriate designated use is of paramount importance in establishing criteria that are appropriately protective of that designated use. In addition, the regulations establish a rebuttable presumption that the uses of protection and propagation of fish, shellfish, and wildlife and recreation in and on the water are attainable and must apply to a water body, unless it has been affirmatively demonstrated that such uses are not attainable.

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[3] For more information about Water Quality Standards, see the WQS Regulation at http://water.epa.gov/lawsregs/lawsguidance/wqs_index.cfm (Accessed February 2016) and EPA's *Water Quality Standards Handbook* at http://water.epa.gov/scitech/swguidance/standards/handbook/ (Accessed February 2016).

Biological assessments can be effectively used to help subcategorize the ALU designations. For example, states may adopt subcategories of a use and set the appropriate criteria to reflect varying needs of such subcategories of uses to differentiate between coldwater and warmwater fisheries (see 40 CFR 131.10(c)). States may also adopt seasonal uses, such as the use of streams or rivers for migratory or spawning purposes (40 CFR 131.10(f)). One major challenge in assigning designated uses for aquatic life to surface waters is separating the natural differences inherent in aquatic ecosystems and appropriately classifying them by type (e.g., naturally coldwater vs. warmwater streams) and location (e.g., ecoregion) from the differences that result from exposure to anthropogenic stressors. Natural or "naturally occurring" conditions can be interpreted as comparable to the range of physical, biological, and chemical conditions observed in undisturbed to minimally disturbed reference sites (Stoddard et al. 2006). When developed using reference data sets from long term biological monitoring and assessment programs, the boundaries for the upper BCG levels can be described in a narrative form and quantified to document the observed natural conditions. The BCG thus provides a descriptive framework to help biologists and water quality managers interpret their aquatic life goals relative to natural conditions. By more fully accounting for natural differences in aquatic ecosystems, designating more specific ALUs helps to reduce a major source of uncertainty and error in water quality management.

The BCG can be used by state programs not only to develop detailed narrative descriptions of ALU goals in terms of the expected biological community, but also to help develop numeric biological criteria for measuring attainment of the goals (USEPA 1990, 2011a). Water quality *criteria* are elements of state WQS expressed as constituent concentrations, levels, or narrative statements representing a quality of water that supports a particular use. When criteria are met, water quality is expected to protect the designated use (40 CFR 131.3). Once adopted into standards, criteria can serve as the basis for (1) controls on point and nonpoint source pollution concentrations to protect aquatic life, (2) statements of expectations for the condition of aquatic life in a water body, and (3) guidelines helpful in water quality planning (e.g., tracking of cumulative loads of point and nonpoint source pollutants). Biological criteria have been defined as narrative expressions or numeric values of the biological characteristics of aquatic communities based on appropriate reference conditions.

1.4.2 Use of the Biological Condition Gradient to Define Levels of Condition

By designating uses and articulating narrative and numeric criteria, states can establish environmental goals for their water resources and measure attainment of these goals. When designating uses, a state may weigh the environmental, social, and economic consequences of different use designations. Water quality regulations allow the state, with public participation, flexibility in weighing these considerations and adjusting designated uses over time. Clearly defining the uses that appropriately reflect the current and potential future uses for a water body, determining the attainability of those goals, and appropriately evaluating the consequences of a designation can be a challenging task.

A principal function of designated uses in WQS is to communicate the desired condition of surface waters to water quality managers, the regulated community, and the public. For designating ALUs, an effective approach is one that readily and transparently translates narrative biological descriptions of the ALU into quantitative measures, such as biological index values. The index values can be adopted into the WQS as biological criteria and thresholds established for assessing attainment. The indices should respond in predictable ways to stress so that degradation can be detected early and incremental improvements tracked. States that have developed robust biological assessment programs typically strive to distinguish different levels of biological condition. States have either made these levels explicit in their WQS by adopting detailed biological descriptions of ALUs, or they have implicitly done so by recognizing levels of condition in their monitoring protocols for assessing attainment of ALU.

Although the benefits of specificity might apply to any of the designated uses described in CWA section 303, the benefits are particularly relevant for ALUs, because a broad range of biological conditions can be interpreted as supporting an ALU. For example, biological conditions in a minimally disturbed stream in a wilderness area would likely support a biotic community close to what would naturally be expected, whereas the biological condition in a stream in a more developed watershed might be measurably impacted relative to the wilderness stream, the degree of impact dependent upon effectiveness of best management practices (BMPs) that have been implemented. Under non-specific ALU classification with a single ALU threshold, both streams might be judged as meeting the designated ALU, and a threshold might be set that does not protect the higher biological conditions in the wilderness stream from degrading. By specifically articulating ALU goals for systems with different levels of human disturbance, deterioration can be detected and preventive management actions can be triggered earlier in the process prior to serious and irretrievable degradation. The BCG provides a framework for defining management goals and designated uses for water bodies having different levels of biological condition.

Chapter 2. The Biological Condition Gradient: Fundamental Concepts

The BCG is a scientific framework that supports more refined interpretation of biological condition even when assessment approaches may differ. The BCG combines scientific knowledge with the practical observations and experience of biological assessment practitioners (Figure 4) with the needs of resource managers. In conjunction with other environmental data and information, it can be used by environmental practitioners to help:

- Determine the environmental conditions that exist, relative to naturally-derived conditions—The BCG provides a common language with which to interpret and communicate current ecological conditions relative to baseline conditions that are anchored in level 1 of the BCG, "as naturally occurs."
- Decide what environmental conditions are desired—The BCG can be used with expert groups and stakeholders to set easily communicated environmental goals.
- Plan for how to achieve these conditions—The BCG provides a scientific basis for planning, restoration, protection, and monitoring by providing a common language and a pathway to shared quantitative goals.

Figure 4. Biologists conducting stream and lake assessments.

The BCG translates the theoretical and empirical work of researchers and practitioners to create a nationally-applicable model that helps to link management goals for resource condition with the quantitative measures used in biological assessments. As discussed in Chapter 1, the conceptual BCG was developed and tested by an expert workgroup that included scientists from 21 states, an interstate basin association, and a tribe. The BCG was designed to describe ecological response to anthropogenic stressors in sufficient detail so that a site can be placed into a level[4] along the BCG continuum through use of the core data elements collected by most state monitoring programs (USEPA 2013a). This framework can be used to organize biological, chemical, physical, and land cover data and information to interpret changes in assemblage composition and structure, spatial and temporal size of disturbance, and declines in function and connectivity relative to a baseline of undisturbed or minimally disturbed conditions.

[4] A full description of the BCG levels is provided in section 2.3.

The BCG provides an interpretative framework explicitly linking science and monitoring information to goals in water quality standards and criteria and, thus, aids in management decision making (Davies and Jackson 2006). Each of the proposed six levels of the BCG is described via a detailed narrative that communicates ecological characteristics associated with that condition level. In this way, the descriptive gradient can be used to interpret numeric metric scores into a fuller understanding of their ecological meaning and importance. Once calibrated to local data, the BCG creates a bridge between biological metric scores and the condition levels with which they are commonly associated.

2.1 The Scientific Foundation of the Biological Condition Gradient

The practice of using biological indicators to assess water quality is over a century old, and the scientific foundation of the BCG is based on many decades of biologists' accumulated experience with biological assessment and monitoring. The Saprobien System is a concept based on organism tolerance proposed by Lauterborn in 1901 and further developed by Kolkwitz and Marsson (Davis 1995). This system uses benthic macroinvertebrates and planktonic plants and animals as indicators of organic loading and low dissolved oxygen (DO). It has been updated since its initial development and is currently used in several European countries. The limnologists Thienemann and Naumann developed the concept of trophic state classification for lakes in the 1920s (Cairns and Pratt 1993; Carlson 1992). Both the Saprobien System and lake trophic state classifications describe a response gradient (or response classes for lakes) to nutrient pollution. The Saprobien System was explicitly developed to assess human pollution in rivers, but the trophic state concept was originally developed to describe natural conditions in lakes and only later became a concept to describe pollution-induced eutrophication (e.g., Vollenweider 1968). The 1950s marked the development of Beck's biotic index in the U.S. and Pantle and Buck's Saprobic Index in Europe, both of which were directly based on the Saprobien System (Beck 1954; Pantle and Buck 1955). The Saprobic Index, which led to the development of the widely used Hilsenhoff Index (e.g., Hilsenhoff 1987a, 1987b) in the U.S., could be considered the predecessor of today's biotic indices (Davis 1995). Later studies used diversity indices based on information theory to describe changes in community structure, richness, and dominance (evenness) as a measure of pollution effects (e.g., Wilhm and Dorris 1966).

Biological information from monitoring programs has been frequently synthesized by constructing biotic indices, such as the IBI (Karr 1981; Karr et al. 1986). The IBI integrates the concept of anchoring the measurement system in undisturbed reference conditions with the measurement of several indicators intended to reflect ecological components of composition, diversity, and ecosystem processes. It thus combines a conceptual model of ecosystem change in response to increasing levels of stressors with a practical measurement system. The BCG is also grounded in the concepts of stress ecology articulated by Odum et al. (1979), Odum (1985), Rapport et al. (1985), and Cairns et al. (1993), describing "natural" conditions and the change in biological condition caused by stressors. To achieve maximum potential application nationwide, the BCG levels were developed based on state biologists' experiences with water quality management (Courtemanch et al. 1989; Yoder and Rankin 1995a), as well as the practical experience of a diverse group of aquatic scientists from different bio-geographic areas (Davies and Jackson 2006). The BCG:

- Describes a scale of six condition levels, from undisturbed (level 1) to highly disturbed conditions (level 6).
- Synthesizes existing field observations and generally accepted interpretations of patterns of biological change within a common framework.

- Incrementally measures how a system may have departed from undisturbed condition, based on observable, ecological attributes.

In its initial development, the description of biological attributes that make up the model applied best to permanent, hard-bottom streams that are exposed to increases in temperature, nutrients, fine sediments, and other pollutants. This is the stream-type and stressor regime originally described by the model and the one most developed to date, for example, in Alabama, Connecticut, Maine, Maryland, Minnesota, New Jersey, Ohio, and Vermont. The model has been further tested with states in different parts of the country and increasingly in different water body types (e.g., headwater streams, coastal plains freshwater streams, rivers, wetlands, estuaries, and coral reefs) to evaluate the national applicability of the model (see Appendix B for examples). Results have shown good correlation with some necessary refinement of the model attributes to accommodate regional and water body differences. For example, for the southern great plains region, attribute II, originally defined as sensitive-rare taxa, was redefined as *highly sensitive taxa* because rarity of a taxon in the region was not associated with sensitivity to stress. In this region, many rare, native taxa might be highly tolerant to stressors, such as low DO and high temperature. Through similar developmental processes, the BCG, as initially developed and tested, is applicable to other aquatic ecosystems and stressors with appropriate modifications. The BCG should be viewed as a scientific framework that can readily incorporate future advances in scientific understanding. The model building was initially based on expert consensus and then further tested and refined following procedures detailed in Chapter 3. Quantitative approaches for translating the narrative model into numeric values are discussed in Chapter 4.

The value of a conceptual framework such as the BCG is not only that it documents experimentally established knowledge, but that it also promotes a more rigorous testing of empirical observations by clearly stating them in a provisional model (Davies and Jackson 2006). Conceptual models formalize the state of knowledge and guide research. Empirically-based generalizations have led to conceptual models that describe the behavior of biological systems under stress (Brinkhurst 1993; Fausch et al. 1990; Karr and Dudley 1981; Margalef 1963, 1981; Odum et al. 1979; Rapport et al. 1985; Schindler 1987). For example, Brinkhurst (1993) observed that "Everyone knew [in 1929] that increases in numbers and species could be related to mild pollution, that moderate pollution could produce changes in taxa so that diversity remained similar but species composition shifted, and that eventually species richness declined abruptly and numbers of some tolerant forms increased dramatically." Such ecosystem responses to stressor gradients have been portrayed as a progression of stages that occur in a generally consistent pattern (Cairns and Pratt 1993; Odum 1985; Odum et al. 1979; Rapport et al. 1985). Establishing and validating quantifiable thresholds along that progression with empirical data is a priority need for resource managers (Cairns 1981).

2.2 The Biological Condition Gradient Attributes

The BCG framework depicts ecological condition in terms of observable or measurable changes in an aquatic system in response to anthropogenic stress. The characteristics, described as "attributes" in this document, were selected because they corresponded to the characteristics used by state workgroup members to measure biological condition and develop biological criteria. The 10 attributes are discussed below and listed in Table 1. In biological assessments, most information is collected at the spatial scale of a site or reach and the temporal scale ranging from a season to as short as a single sampling event. Many of the attributes that make up the BCG are based on these scales. Site scale attributes include aspects of taxonomic composition and community structure (attributes I–V), organism condition (attribute VI), and organism and system performance (attributes VII and VIII). At larger temporal and spatial scales, physical-biotic interactions (attributes IX and X) are also included because of their importance to state water quality management programs in evaluating longer-term impacts, determining restoration potential, and tracking recovery in specific water bodies.

Information used to assess the ten attributes may be acquired from two sources. Sample-based data from instream monitoring using standardized sampling protocols can produce the most reliable, reproducible form of information and are best used for attributes II–V. Knowledge-based information, such as evidence from natural history surveys, agency records and reports (e.g., stocking reports), academic studies and journal publications, expert observations, and so on, can contribute significantly to BCG development even when methods are inconsistent. Since many of the attributes rely on the positive observation (i.e., presence) of an organism and its relative occurrence in the community, any reliable sources of information can be used to develop and calibrate the BCG for a specific water body and/or region. Attributes I–X are described below (from Davies and Jackson 2006).

Attribute I: Historically Documented, Sensitive, Long-lived, or Regionally Endemic Taxa

Attribute I can be developed using both sample-based and knowledge-based sources. Taxa that are *historically documented* refer to those known to have been supported in a water body or region according to historical records. This attribute was derived to cover taxa that are s*ensitive or regionally endemic* that have restricted, geographically isolated distribution patterns (occurring only in a locale as opposed to a region), often due to unique life history requirements. They may be long-lived and late maturing and have low fecundity, limited mobility, multiple habitat requirements as with diadromous species, or require a mutualistic relationship with other species. They may be among listed Endangered or Threatened (E/T) or special concern species. Predictability of occurrence is often low, and therefore requires documented observation. The presence or absence of a population might provide significant information in an assessment, but there are typically insufficient data to develop the stress response relationships needed to assign these taxa to attributes II through V (as discussed below). Recorded occurrence may be highly dependent on sample methods, site selection, and level of effort, thus requiring use of knowledge-based sources in addition to actual instream sampling. The taxa that are assigned to this category require expert knowledge of life history and regional occurrence of the taxa to appropriately interpret the significance of their presence or absence. Long-lived species are especially important as they provide evidence of multi-annual persistence of habitat condition. For example, many species of freshwater mussels in the Southeast U.S. are highly endemic and have been extirpated in many areas. The presence of freshwater mussels in a stream might signify high quality conditions, but their absence does not necessarily indicate poor conditions if overharvesting of the mussels is the cause.

Attribute II: Highly Sensitive Taxa

Highly sensitive taxa typically occur in low numbers relative to total population density, but they might make up a large relative proportion of richness. In high quality sites, they might be ubiquitous in occurrence or might be restricted to certain micro-habitats. Many of these species commonly occur at low densities, so their occurrence is dependent on sample effort. They are often stenothermic (i.e., having a narrow range of thermal tolerance) or cold-water obligates, and their populations are maintained at a fairly constant level, with slower development and a longer life-span. They might have specialized food resource needs, feeding strategies, or life history requirements, and they are generally intolerant to significant alteration of the physical or chemical environment. They are often the first taxa lost from a community following moderate disturbance or pollution.

In earlier descriptions of the BCG, highly sensitive taxa were called *sensitive-rare* taxa (Davies and Jackson 2006), but experience with calibrating the BCG showed that some highly sensitive species are found at many exceptional sites, and some were occasionally highly abundant (e.g., Snook et al. 2007). The distinguishing characteristic for this attribute category was found to be sensitivity and not relative rarity, although some of these taxa might be uncommon in the data set (e.g., very small percent of sample occurrence or sample density)

Attribute III: Intermediate Sensitive Taxa

Intermediate sensitive taxa were formerly labeled sensitive-ubiquitous taxa (Davies and Jackson 2006), but subsequent development revealed that the experts relied upon the sensitivity of a species to stress rather than whether it was "ubiquitous," though intermediate sensitive taxa are ordinarily common and abundant in natural communities. They tend to have a broader range of tolerances than highly sensitive taxa, and they usually occur in reduced abundance and reduced frequencies at disturbed or polluted sites. These taxa often comprise a substantial portion of natural communities.

Attribute IV: Intermediate Tolerant Taxa

Attribute IV taxa commonly comprise a substantial portion of an assemblage in undisturbed habitats, as well as in moderately disturbed or polluted habitats. They exhibit physiological or life-history characteristics that enable them to thrive under a broad range of thermal, flow, or oxygen conditions. Many have generalist or facultative feeding strategies enabling utilization of diverse food types. These species have little or no detectable response to moderate stress, and they are often equally abundant in both reference and moderately stressed sites. Some intermediate tolerant taxa may show an "intermediate disturbance" response, where densities and frequency of occurrence are relatively high at intermediate levels of stress, but they are intolerant of excessive pollution loads or habitat alteration.

Attribute V: Tolerant taxa

Tolerant taxa are those that typically comprise a low proportion of natural communities. These taxa are more tolerant of a greater degree of disturbance and stress than other organisms and are, thus, resistant to a variety of pollution or habitat induced stress. They may increase in number (sometimes greatly) under severely altered or stressed conditions. They may possess adaptations in response to organic pollution, hypoxia, or toxic substances. These are the last survivors in severely disturbed systems and can prevail in great numbers due to lack of competition or predation by less tolerant organisms, and they are key community components of level 5 and 6 conditions.

Attribute VI: Non-native or Intentionally Introduced Taxa

With respect to a particular ecosystem, species fitting attribute VI are any species not native to that ecosystem. Species introduced or spread from one region of the U.S. to another outside their normal ranges are non-native, or non-indigenous. This category also includes species introduced from other continents and referred to as "alien" species. Attribute VI can also include introduced disease or parasitic organisms. This attribute represents both an effect of human activities and a stressor in the form of biological pollution. Although some intentionally introduced species are valued by large segments of society (e.g., gamefish), these species might be as disruptive to native species as undesirable opportunistic invaders (e.g., zebra mussels). Many rivers in the U.S. are dominated by non-native fish and invertebrates (Moyle 1986), and the introduction of non-native species is the second most important factor contributing to fish extinctions in North America (Miller et al. 1989). The BCG identifies the presence of native taxa expected under undisturbed or minimally disturbed conditions as an essential characteristic of BCG level 1 and 2 conditions. The BCG only allows for the occurrence of non-native taxa in these levels if those taxa do not displace native taxa and do not have a detrimental effect on native structure and function. Condition levels 3 and 4 depict increasing occurrence of non-native taxa. Extensive replacement of native taxa by tolerant or invasive, non-native taxa can occur in levels 5 and 6. Attribute VI may rely on either sample-based or knowledge-based sources.

Attribute VII: Organism Condition

Organism condition is an element of ecosystem function, expressed at the level of anatomical or physiological characteristics of individual organisms. Organism condition includes direct and indirect indicators such as fecundity, morbidity, mortality, growth rates, and anomalies (e.g., lesions, tumors, and deformities). Some of these indicators are readily observed in the field and laboratory, whereas the assessment of others requires specialized expertise and much greater effort. Organism condition can also change with season or life stage, or occur as short-term events making assessment difficult. The most common approach for state programs is to forego complex and demanding direct measures of organism condition (e.g., fecundity, morbidity, mortality, disease, growth rates) in favor of indirect or surrogate measures (e.g., percent of organisms with anomalies, age or size class distributions) (Simon 2003). Organism anomalies in the BCG vary from naturally occurring incidence in levels 1 and 2 to higher than expected incidence in levels 3 and 4. In levels 5 and 6, biomass is reduced, the age structure of populations indicates premature mortality or unsuccessful reproduction, and the incidence of serious anomalies is high. This attribute has been successfully used in stream indices based on the fish assemblage (Sanders et al. 1999; Yoder and Rankin 1995a). Incidence of disease is being evaluated as an indicator of organism condition for the coral reef BCG (see Appendix B-3).

Attribute VIII: Ecosystem Function

Ecosystem function refers to any processes required for the performance of a biological system expected under naturally occurring conditions. Naturally occurring conditions have been typically interpreted as those conditions found in undisturbed to minimally disturbed conditions but some processes can be sustained under moderate levels of disturbance. Examples of ecosystem functional processes are primary and secondary production, respiration, nutrient cycling, and decomposition. Assessing ecosystem function includes consideration of the aggregate performance of dynamic interactions within an ecosystem, such as the interactions among taxa (e.g., food web dynamics) and energy and nutrient processing rates (e.g., energy and nutrient dynamics) (Cairns 1977).

Additionally, ecosystem function includes aspects of all levels of biological organization (e.g., individual, population, and community condition). Altered interactions between individual organisms and their abiotic and biotic environments might generate changes in growth rates, reproductive success, movement, or mortality. These altered interactions are ultimately expressed at ecosystem-levels of organization (e.g., shifts from heterotrophy to autotrophy, onset of eutrophic conditions) and as changes in ecosystem process rates (e.g., photosynthesis, respiration, production, decomposition).

At this time, the level of effort required to directly assess ecosystem function is beyond the means of most state monitoring programs. Instead, in streams and wadeable rivers, most programs rely on taxonomic and structural indicators to make inferences about functional status (Karr et al. 1986). For example, shifts in the primary source of food might cause changes in trophic guild indices or indicator species. Although direct measures of ecosystem function are currently difficult or time consuming, they might become practical in the future (Gessner and Chauvet 2002). The BCG conceptual model includes ecosystem function for future application.

Attribute IX: Spatial and Temporal Extent of Detrimental Effects

The spatial and temporal extent of stressor effects includes the near-field to far-field range of observable effects of the stressors on a water body. Such information can be conveyed by biological assessments provided the spatial density of sampling sites is sufficient to convey changes along a pollution continuum (USEPA 2013a). Use of a continuum provides a method for determining the severity (i.e., departure from the desired state) and extent (i.e., distance over which adverse effects are observed) of an impairment from one or more sources. Yoder et al. (2005) detailed this approach in their historical assessment of large rivers in Ohio. As with attribute VIII above, attribute IX has not yet been developed and applied in BCG models for specific streams and wadeable rivers. It is included for future development and application. State scientists involved in the development of the BCG conceptual model stated that this attribute was important to include for future testing and development. Some state biological monitoring and assessment programs document the spatial and temporal extent of stressor effects and use the information to predict the recovery potential of a degraded stream, as well as the risk of degradation in high quality streams. This information informs water quality management decisions on prioritization of actions. The National Hydrography Dataset (NHD) (USGS 2014), together with biological assessment information from attributes I–VIII can be an important tool to help evaluate position and extent of condition and stressors in a water body or watershed by mapping the locations (i.e., spatial distribution) of the biological samples.

Attribute X: Ecosystem Connectance

Attribute X refers to the access or linkage (in space/time) to materials, locations, and conditions required for maintenance of interacting populations of aquatic life. It is the opposite of fragmentation and is necessary for persistence of metapopulations and natural flows of energy and nutrients across ecosystem boundaries. Ecosystem connectance can be indirectly expressed by certain species that depend on the connectance, or lack of connectance, within an aquatic ecosystem to fully complete their life cycles and thus maintain their populations. Diadromous fish species are one such example—their absence or presence can provide information on the presence or absence of critical habitats to support different life stages. However, the inverse of connectance, isolation, is important for some species (e.g., amphibians, which are negatively impacted by fish that gain access to amphibian habitat via artificial or natural connections). This difference dependence upon connectance underscores the importance of well-defined BCG levels 1 and 2 as the benchmark for interpreting change in the BCG attributes. The NHD can be an important tool to evaluate the extent of connections (or occurrence of barriers or habitat disconnects) in a water body or watershed. A habitat mosaic measure is being evaluated as an indicator of ecosystem connectance in the estuarine BCG (see Appendix B-2).

2.3 The Biological Condition Gradient Levels of Biological Condition

The BCG has been divided into six levels along a generalized stressor-response continuum to provide discrimination of different levels of condition that are detectable, given current assessment methods and well-designed monitoring protocols. Since the BCG is a continuum, in principle it is possible to determine more or fewer levels depending upon the discriminatory power of a state water quality management program (USEPA 2013a). The six levels are proposed as a hypothetical framework for which the practical concerns of the state would determine the number of levels that can be implemented. For example, in most forested perennial stream ecosystems it may be technically possible to discriminate six classes in the condition gradient, ranging from undisturbed to highly disturbed conditions (Davies and Jackson 2006). However, some states or regions may only be capable of discriminating two or three levels, given current technical program capabilities, while others might be capable of discerning six or more levels based on highly proficient programs and robust data sets (USEPA

2013a). In addition, some regions of the country may not currently support level 1 water conditions. *Regardless of the number of levels a state can detect, the BCG framework is to be a starting point for a state to think about how to use biological information to better determine existing conditions and potential for improvement and how to use the information to better communicate biological condition and to set water quality objectives*.

The six levels of the BCG are described as follows (modified from Davies and Jackson 2006).

- **Level 1, Natural or native condition**—*Native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability.* Level 1 represents biological conditions as they existed (or still exist) in the absence of measurable effects of stressors and provides the basis for comparison to the next five levels. The level 1 biological assemblages that occur in a given biogeophysical setting are the result of adaptive evolutionary processes and biogeography. For this reason, the expected level 1 assemblage of a stream from the arid southwest will be very different from that of a stream in the northern temperate forest. The maintenance of native species populations and the expected natural diversity of species are essential for levels 1 and 2. Non-native taxa (attribute VI) might be present in level 1 if they cause no displacement of native taxa, although the practical uncertainties of this provision are acknowledged (see section 2.2). Attributes I and II (i.e., historically documented and sensitive taxa) can be used to help assess the status of native taxa when classifying a site or assessing its condition.
- **Level 2, Minimal changes in structure of the biotic community and minimal changes in ecosystem function**—*Most native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability.* Level 2 represents the earliest changes in densities, species composition, and biomass that occur as a result of slight elevation in stressors (e.g., increased temperature regime or nutrient pollution). There might be some reduction of a small fraction of highly sensitive or specialized taxa (attribute II) or loss of some endemic or rare taxa as a result. The occurrence of non-native taxa should not measurably alter the natural structure and function and should not replace any native taxa. Level 2 can be characterized as the first change in condition from natural, and it is most often manifested in nutrient-polluted waters as slightly increased richness and density of either intermediate sensitive and intermediate tolerant taxa (attributes III and IV) or both. These early response signals have been observed in many state programs as illustrated in Figure 5, which shows slight to moderate increases of mayfly density in response to increases in conductivity in Maine streams. Mayfly taxa typically have been identified in Maine as sensitive ubiquitous taxa and show an increase to initial levels of some stress (e.g., an increase in conductivity or nutrient pollution), followed by a decrease in abundance as stress levels continue to rise.

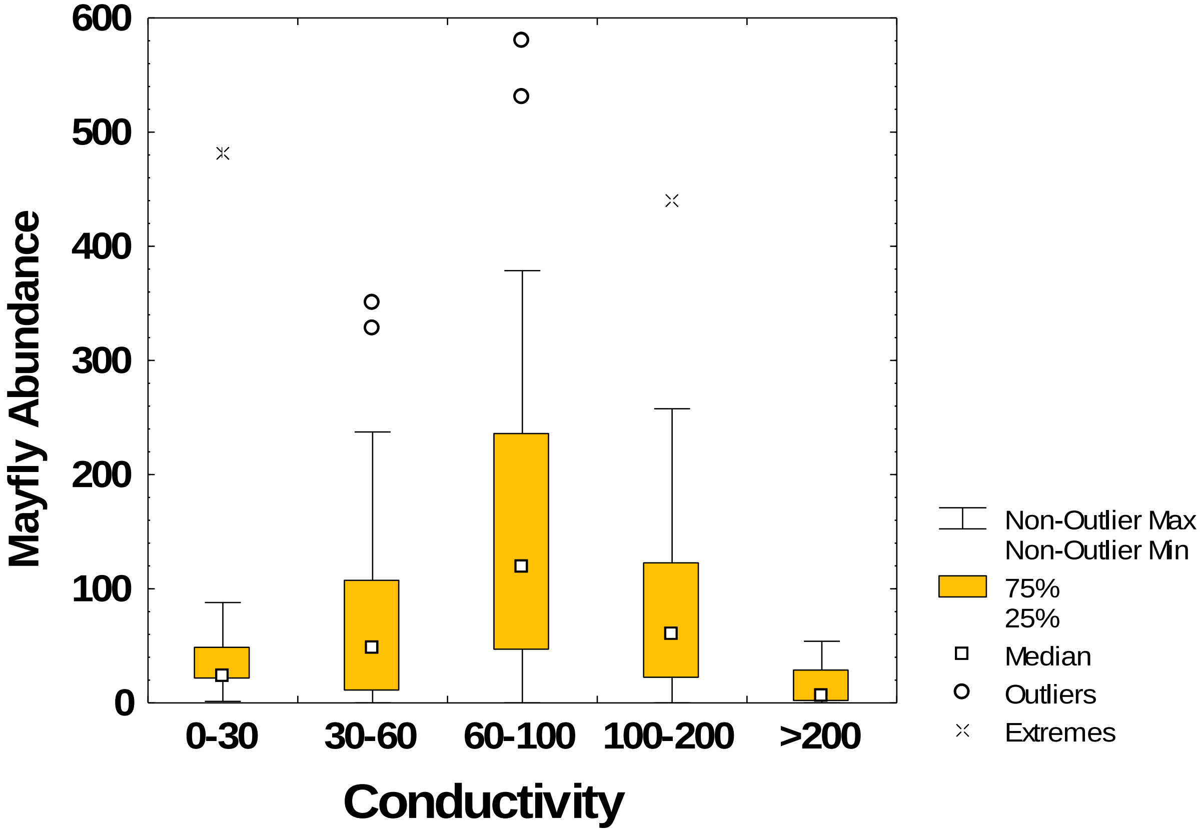

Figure 5. Response of mayfly density to stress in Maine streams as indicated by a gradient of increasing conductivity.

- **Level 3, Evident changes in structure of the biotic community and minimal changes in ecosystem function**—*Evident changes in structure due to loss of some highly sensitive native taxa; shifts in relative abundance of taxa, but sensitive-ubiquitous taxa are common and relatively abundant; ecosystem functions are fully maintained through redundant attributes of the system.* Level 3 represents readily observable changes that, for example, can occur in response to organic pollution or increased temperature. The "evident" change in structure for level 3 is interpreted to be perceptible and detectable decreases in highly sensitive taxa (attribute II), and increases in sensitive-ubiquitous taxa or intermediate organisms (attributes III and IV). Attribute IV taxa (intermediate intolerance) might increase in abundance as an opportunistic response to nutrient or organic inputs.
- **Level 4, Moderate changes in structure of the biotic community with minimal changes in ecosystem function**—*Moderate changes in structure due to replacement of some intermediate sensitive taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes.* Moderate changes of structure occur as stressor effects increase in level 4. A substantial reduction of the two sensitive attribute groups (attributes II and III) and replacement by more tolerant taxa (attributes IV and V) might be observed. A key consideration is that some attribute III sensitive taxa are maintained at a reduced level, but they are still an important functional part of the system (i.e., function is

maintained). While total abundance (density) of organisms might increase, no single taxa or functional group should be overly dominant.

- **Level 5, Major changes in structure of the biotic community and moderate changes in ecosystem function**—*Sensitive taxa are markedly diminished or missing; conspicuously unbalanced distribution of major groups from those expected; organism condition shows signs of physiological stress; ecosystem function shows reduced complexity and redundancy; increased build-up or export of unused materials.* Changes in ecosystem function (as indicated by marked changes in food-web structure and guilds) are critical in distinguishing between levels 4 and 5. This could include the loss of functionally important sensitive taxa and keystone taxa (attribute I, II, and III taxa), such that they are no longer important players in the system, though a few individuals may be present. Keystone taxa control species composition and trophic interactions, and are often, but not always, top predators. As an example, removal of keystone taxa by overfishing has greatly altered the structure and function of many coastal ocean ecosystems (Jackson et al. 2001). Additionally, tolerant non-native taxa (attribute VI) may dominate some assemblages, and changes in organism condition (attribute VII) may include significantly increased mortality, depressed fecundity, and/or increased frequency of lesions, tumors, and deformities.
- **Level 6, Severe changes in structure of the biotic community and major loss of ecosystem function**—*Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered.* Level 6 systems are taxonomically depauperate (i.e., low diversity and/or reduced number of organisms) compared to the other levels. For example, extremely high or low densities of organisms caused by excessive organic pollution, severe toxicity, and/or severe habitat alteration may characterize level 6 systems. Non-native taxa may predominate.

2.3.1 Bringing the Biological Condition Gradient Levels and Attributes Together

The BCG narrative portrays general patterns of biological and ecological response common across regions, as measured by the BCG attributes. Table 2 organizes the ten BCG attributes into six categories: community structure, non-natives, condition, function, landscape, and connectivity. Attributes I through V have been combined in one category in Table 2—structure and compositional complexity. This category typically includes measures of the number, type, and proportion of individual taxa within an assemblage (e.g., benthic macroinvertebrates, fish, and algal assemblages). These attributes are the foundation of most state biological assessment programs for streams and wadeable rivers. The five taxonomic attributes characterize biological sensitivity to the cumulative impact of stressors (e.g., highly, intermediate, or tolerant taxa). In addition to the sensitivity-based attributes, biologists have also used assemblage richness and balance, assemblage abundance or biomass, and keystone or habitat-structuring species (e.g., reef-building corals) to define attributes and distinguish levels of condition along a stress gradient. Attributes respond to stressors in distinctly different ways so that there are predictive, quantitative measures along the full range of stress levels (Figure 6, Table 3). Defining and quantifying these changes along the full gradient of stress effects is necessary for developing reliable, predictable measures for incremental changes in biological condition. For example, highly sensitive taxa might disappear from a community in early, or low, levels of stress. Tolerant taxa might become more dominant as stress increases, not only because they might thrive, but also because there are fewer sensitive species and the proportion of tolerant taxa in the entire community increases. Intermediate tolerant taxa might not provide a significant signal under most conditions if they are present under a wide range of stress. However, the absence of this group of taxa in highly stressed conditions can help document highly disturbed conditions, and their reappearance may indicate initial response to management actions for restoration. As work proceeds on applying the BCG to other water body types and developing approaches for including additional assemblages (e.g., periphyton, amphibians, birds) and new methods for sampling and analyzing aquatic life (e.g., DNA analysis), it is expected that these attributes will be refined and comparable detailed descriptions for the remaining attributes will emerge.

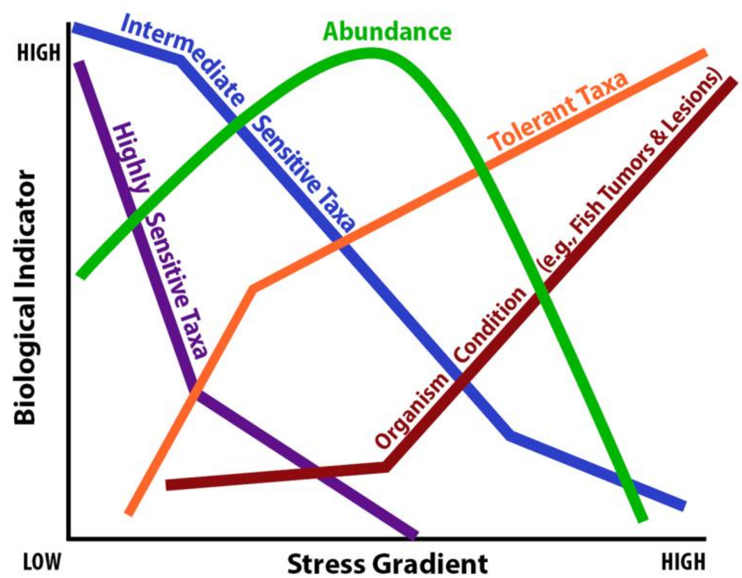

Figure 6. Hypothetical examples of biological response to the cumulative impact of multiple stressors.

Table 2. BCG: Ecological Attributes

	Attribute Grouping	Description	Examples of BCG					
			1	2	3	4	5	6
STRUCTURE	Structure and Compositional Complexity (Attributes I–V) See Table 3 for detailed descriptions for these attributes.	Community or habitat structure and complexity. May also recognize loss of habitats or species due to human activities. Examples include macroinvertebrate or fish indices, phytoplankton or zooplankton community measures, epifaunal measures, biotope mosaics, presence/quantity of sensitive taxa or biotopes, wetland vegetative indices, etc.	Community composition is as naturally occurs, except for global extinctions based on observations from water bodies with similar habitat and ecoregion without measurable human-caused stressors (this includes chlorophyll a levels, biotope mosaics, species composition including large, long-lived, and sensitive species; patterns of vegetation are as naturally occurs)	Minor changes in natural occurrences of biotopes or patterns of vegetation, slight decreases in sensitive species, and slight increases in tolerant species	Evident changes in biological metrics (decreases in sensitive species and increases in more tolerant species, evident changes in vegetation patterns); may be slight decreases in biotope or habitat area; biotope mosaic basically intact	Significant changes in biological metrics (marked decreases in sensitive species [including large or long-lived taxa] and increases in tolerant species, evident changes in vegetation patterns); biotope mosaic slightly altered with replacement of natural habitats/biotopes with tolerant or non-naturally occurring components; detectable loss in some biotope types or habitat area	Most sensitive, large and/or long-lived taxa are absent, with a dominance in abundance of tolerant taxa; significant shifts in species diversity, size, and densities of remaining species; biotope mosaic significantly altered with many natural habitats/biotopes lost with replacement by tolerant or non-naturally occurring components; evident loss in biotope or habitat area	Sensitive, large, and/or long-lived taxa largely absent; possible high or low extremes in abundance of remaining taxa; marked reduction in species diversity and in size spectra of remaining organisms; near complete loss or alteration of natural biotope mosaic with marked loss in biotope or habitat area

	Attribute Grouping	Description	Examples of BCG					
			1	2	3	4	5	6
NON-NATIVES	Non-Native Taxa (Attribute VI)	Status of non-native species. May include measures of the impact of invasive and non-native species. Examples include estimated numbers of species or individuals, relative density or biomass measures of natives and non-natives, or replacement of native species	Non-native taxa, if present, do not significantly reduce native taxa or alter structural or functional integrity	Non-native taxa may be present, but occurrence has a non-detrimental effect on native taxa	Non-native taxa may be prominent in some assemblages (e.g., crustaceans, bivalves, fish) and some sensitive native taxa may be reduced or replaced by equivalent non-native species (e.g., replacement of native trout with introduced salmonids)	Increased abundance of tolerant non-native species (e.g., Common Carp, non-native centrarchids, Common Reed) or native species (e.g., salmonids) only maintained by regular stocking	Some assemblages (e.g., mollusks, fishes, macrophytes) are dominated by invasive non-native taxa (e.g., Silver Carp, Zebra Mussels, Eurasian Watermilfoil); or increasing dominance by tolerant non-native species such as Common Carp	Same as level 5; not distinguishable based on non-native species alone
CONDITION	Organism Condition (Attribute VII)	Measures condition of individual organisms, including anomalies and diseases. Examples include external anomalies, lesions, disease outbreaks (local or widespread), coral bleaching, seagrass condition, fish pathology, and frequency of diseased or affected organisms	Diseases and anomalies are consistent with naturally occurring incidents and characteristics	Diseases and anomalies are consistent with naturally occurring incidents and characteristics	Incidences of diseases and anomalies may be slightly higher than expected conditions	Incidences of diseases and anomalies are slightly higher than expected. For example, coral bleaching events may occur sporadically and result in slightly elevated mortality. Anomalies in fish occur in a small fraction of a population	Disease outbreaks are increasingly common, anomalies are increasingly common, particularly in long-lived taxa where biomass may also be reduced (e.g., bleaching events are frequent enough to cause mortality of corals). Anomalies, such as deformities, erosion, lesions, and tumors in fish, occur in a measurable fraction of a population	Host species in which diseases and anomalies have been observed are now absent, so diseases might be difficult to detect. Anomalies, disease, etc. may occur across multiple species or taxa groups

	Attribute Grouping	Description	Examples of BCG					
			1	2	3	4	5	6
FUNCTION	Function (Attribute VIII)	Measures of energy flow, trophic linkages and material cycling. They may include proxy or snapshot structural metrics that correlate to functional measures. Examples include photosynthesis: respiration ratios, benthic: pelagic production rates, chlorophyll a concentrations, macroalgal biomass, bacterial biomass and activity	Energy flows, material cycling, and other functions are as naturally occur; characterized by complex interactions and long-lived links supporting large, long-lived organisms	Energy flows, material cycling, and other functions are within the natural range of variability; characterized by complex interactions and long-lived links supporting large, long-lived organisms	Virtually all functions are maintained through operationally redundant system attributes, minimal changes to export and other indicative functions. Some functions increased due to pollution or low level disturbance (e.g., production, biomass, respiration)	Most functions are maintained through operationally redundant system attributes, though there is evidence of loss of efficiency (e.g., increased export or decreased import, there may be shifts in benthic: pelagic production rates	Loss of some ecosystem functions are manifested as changed export or import of some resources and changes in energy exchange rates (photosynthesis: respiration ratios, benthic: pelagic production rates, respiration or decomposition rates)	Most functions show extensive and persistent disruption, shifts to primary production, microbial dominance, fewer and shorter-length trophic links and highly simplified trophic structure, marked shifts in benthic: pelagic production rates
LANDSCAPE	Spatial and Temporal Extent of Detrimental Effects (Attribute IX)	Measures of a landscape's capacity, contributing surface water to a single location, to maintain the full range of ecological processes and function that support a resilient, naturally occurring aquatic community. The functions and processes to be measured include hydrologic regulation, regulation of water chemistry and sediments, hydrologic connectivity (see also attribute X), temperature regulation, and habitat provision	N/A—A natural disturbance regime is maintained	Limited to small pockets and short duration	Limited to a local area or within a season	Mild detrimental effects may be detectable beyond the local area and may include more than one season	Detrimental effects extend far beyond the local area leaving only a few islands of adequate conditions; effect extends across multiple seasons	Detrimental effects may eliminate all refugia and colonization sources within a region or catchment and affect multiple seasons

	Attribute Grouping	Description	Examples of BCG					
			1	2	3	4	5	6
CONNECTIVITY	Ecosystem Connectance (Attribute X)	Observations of exchange or migrations of biota between adjacent water bodies or habitats. Important measures within the area being studied may be strongly affected by factors adjacent to or larger than the immediate study area. Metrics may include dams, causeways, fragmentation measures, hydrological measures, or proxies such as characteristic migratory species	System is naturally connected, or disconnected, in space and time, exchanges, migrations, and recruitment from adjacent water bodies or habitats are as naturally occurs	System is naturally connected, or disconnected, in space and time, exchanges, migrations, and recruitment from adjacent water bodies or habitats are as naturally occurs	Slight loss, or increase, in connectivity between adjacent water bodies or habitats (e.g., between upstream and downstream water bodies), but colonization sources, refugia, and other mechanisms mostly compensate. May also be increase in connectivity due to canals, interbasin transfers	Some loss, or increase, in connectivity between adjacent water bodies or habitats (e.g., between upstream and downstream water bodies), but colonization sources, refugia, and other mechanisms prevent complete disconnects or other failures	Significant loss, or increase, in ecosystem connectivity between adjacent water bodies or habitats (e.g., between upstream and downstream water bodies or habitats) is evident; recolonization sources do not exist for some taxa, some near-complete disconnects or connect exist	For many groups, a complete loss in ecosystem connectivity in at least one dimension (either spatially or temporally) lowers reproductive or recruitment success or prevents migration or exchanges with adjacent water bodies or habitats, frequent disconnects or other failures. For other groups, a complete loss in ecosystem disconnect in at least one dimension lowers reproductive or recruitment success (e.g., predation of amphibians by fish in once isolated headwater streams)

Table 3. BCG Matrix: Taxonomic Composition and Structure Attributes I–V

	BCG Levels					
Ecological Attributes	1 Natural or native condition	2 Minimal changes in the structure of the biotic community and minimal changes in ecosystem function	3 Evident changes in structure of the biotic community and minimal changes in ecosystem function	4 Moderate changes in structure of the biotic community and minimal changes in ecosystem function	5 Major changes in structure of the biotic community and moderate changes in ecosystem function	6 Severe changes in structure of the biotic community and major loss of ecosystem function
	Native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability	Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability	Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but sensitive-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system	Moderate changes in structure due to replacement of some sensitive-ubiquitous taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes	Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build- up or export of unused materials	Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered
I Historically documented, sensitive, long-lived or regionally endemic taxa	As predicted for natural occurrence except for global extinctions	As predicted for natural occurrence except for global extinctions	Some may be marginally present or absent due to global extinction or local extirpation	Some may be marginally present or absent due to global, regional, or local extirpation	Usually absent	Absent
II Highly sensitive taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Most are maintained with some changes in densities	Some loss, with replacement by functionally equivalent sensitive-ubiquitous taxa	May be markedly diminished	Usually absent or only scarce individuals	Absent
III Intermediate sensitive taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Present and may be increasingly abundant	Common and abundant; relative abundance greater than sensitive-rare, taxa	Present with reproducing populations maintained; some replacement by functionally equivalent taxa of intermediate tolerance.	Frequently absent or markedly diminished	Absent
IV Intermediate tolerant taxa	As predicted for natural occurrence, with at most minor changes from natural densities	As naturally present with slight increases in abundance	Often evident increases in abundance	Common and often abundant; relative abundance may be greater than sensitive- ubiquitous taxa	Often exhibit excessive dominance	May occur in extremely high or extremely low densities; richness of all taxa is low
V Tolerant taxa	As naturally occur, with at most minor changes from natural densities	As naturally present with slight increases in abundance	May be increases in abundance of functionally diverse tolerant taxa	May be common but do not exhibit significant dominance	Often occur in high densities and may be dominant	Usually comprise the majority of the assemblage; often extreme departures from normal densities (high or low)

2.4 How the Conceptual Biological Condition Gradient was Developed, Tested, and Evaluated

The conceptual BCG model was developed and tested by an expert workgroup primarily composed of scientists from government and the research community (Davies and Jackson 2006). This section summarizes how the BCG conceptual model was tested to the satisfaction of the expert workgroup and peer reviewers (from Davies and Jackson 2006). For examples on constructing BCG models and quantitative decision rules applied to specific assemblages and habitats, please see Chapters 3 and 4.

A matrix was created that summarized biologists' experience and knowledge about how biological attributes change in response to stress in aquatic ecosystems (Davies and Jackson 2006). In building the model, the workgroup followed an iterative, inductive approach, similar to means-end analysis (Martinez 1998). The workgroup understood that the primary value of the model is as a tool for shared learning and as a framework for communication.

The workgroup began by testing whether biologists from different parts of the country would draw similar conclusions regarding the condition of a water body using simple lists of organisms and their counts. This approach was initially based on Maine's experience, in which expert biologists independently assigned samples of macroinvertebrates to *a priori* defined levels of biological condition defined by differences in assemblage attributes (Davies et al. 1995; Davies et al. In press; Shelton and Blocksom 2004).

To provide a functional framework for practitioners, the workgroup described how each of the 10 attributes varies across six levels of biological condition along a gradient of increasing anthropogenic stress (i.e., human disturbance). The general model was then described in terms of the biota of a specific region (Maine). Based on 20 years of monitoring data, the Maine BCG describes how the relative densities of specific taxa, with varying sensitivities to stress, change across the BCG levels of condition (Davies and Jackson 2006).

To test the general applicability of the BCG to sampling data taken from other stream systems across the country, the workgroup evaluated how consistently individual biologists classified samples of aquatic biota based on the attributes incorporated into the BCG. Government, field, and research biologists participated in the data exercise. The full workgroup was divided into breakout groups according to region (northeast, south-central, northwest, arid southwest/great plains) and assemblage (fish or invertebrates) expertise. Samples were selected from invertebrate and fish data sets to span as many of the BCG levels as possible (i.e., to span the full gradient of conditions). The invertebrate samples and fish samples used in the tests were collected from six different regions within the U.S. (northeast, mid-Atlantic, southeast, northwest, southwest, central) and included only basic descriptors of stream physical characteristics (e.g., substrate, velocity, width, depth), taxonomic names, densities, and in some cases, metric values. These data represent the basic core elements common to nearly all biological monitoring programs. Participants were asked to place each sample into one of the six condition levels, and they were cautioned not to apply a simple relative quality ranking since all six levels did not necessarily occur within the data sets. Biologists relied primarily on differences in relative abundances and sensitivities of taxa (i.e., attributes I–VI) to make level assignments, because this was the information typically collected in state monitoring programs and the data needed to evaluate the status of the other attributes were not available. Percent concurrence among the individuals was calculated to assess the level of agreement among biologists when applying the BCG to raw data. Perfect concurrence was set to equal the product of the number of raters by the number of streams.

In the first stage of the data exercise, between-biologist differences were evaluated by asking all workgroup participants to rate a single data set of 6–8 samples. The breakout groups were then asked to classify samples from larger and more variable data sets. The groups were also instructed to summarize their interpretations and to identify biological responses to changes in conditions not captured by the BCG. Finally, the workgroup participants identified how, from their perspectives, the BCG levels corresponded to the CWA biological integrity objective and interim goal for protection of aquatic life (e.g., protection and propagation of fish, shellfish, and wildlife).

Overall, workgroup members independently agreed on placement of sites in the same BCG levels for 82% of the benthic macroinvertebrate samples and 74% of the fish samples. When assignments differed, the range of variation among workgroup members was within one level in either direction for all samples with a few exceptions. BCG levels were revised following full workgroup discussion so that transitions were more distinct.

Each of the breakout groups independently reported that the ecological characteristics corresponding to BCG levels 1, 2, 3 and either some or all of BCG level 4 characteristics were generally compatible with how they assess the CWA's interim goal for protection of aquatic life. The experts unanimously agreed that BCG levels 1, 2, and 3 attained the CWA goal and BCG levels 5 and 6 did not. Opinions differed among the experts on whether all or some aspects of BCG level 4 characteristics were compatible with attaining this goal. For example, the workgroup extensively discussed what constituted an acceptable degree of replacement of sensitive taxa by tolerant taxa. However, experts united in a clear consensus that the BCG process provided detailed, readily transparent documentation of the expert logic and underlying science for establishing BCG levels. Additionally, expert discussion on implementation of the BCG framework to interpretation of condition included the following programmatic considerations:

- *The technical rigor of the monitoring program that produced the condition assessments*—Conceptually, a less rigorous monitoring program produces assessments with a greater degree of uncertainty, or precision, and potentially lower accuracy. In lieu of improving the program's technical rigor, or to compensate for uncertainty associated with monitoring programs of lower technical rigor, some experts recommended that a more protective, e.g., conservative, BCG level be used to measure attainment of the CWA ALU goal.
- *Protection of high quality conditions*—The experts identified the characteristics described by BCG levels 1 and 2 as consistent with their understanding of the CWA "biological integrity" objective. Concern was expressed that a single threshold comparable to BCG level 4 is not protective of high ecological quality and that water bodies comparable to BCG levels 1, 2, or 3 would likely decline significantly before action would be triggered to address sources of degradation. Experts noted that restoration and remediation costs are typically much higher than costs for prevention. Experts recommended that multiple thresholds protective of existing ALU conditions be established (e.g., thresholds comparable to BCG levels 2, 3, or 4). Alternatively, if only a single threshold is established, some experts recommended that the threshold should be protective of higher level conditions comparable to BCG level 3.

Workgroup members reported that key concepts were important with respect to classifying samples into levels and identifying the boundaries in between. For levels 1 and 2, biologists identified the maintenance of native species populations as essential to their understanding of biological integrity. Although many participants noted that methods for distinguishing differences between levels in attribute VIII (ecosystem function) were poorly defined, most nevertheless identified ecosystem function changes (as inferred by marked changes in food-web structure and guilds) as critical in distinguishing between levels 4 and 5.

Discussion following the data exercise revealed that participants readily agreed on some of the BCG attributes, but not others. For example, participants indicated they mostly used attributes I–V (taxonomic composition, pollution sensitivity), attribute VI (non-native taxa, for levels 2–6 only), and attribute VII (organism condition) to evaluate biological conditions in streams and wadeable rivers. In contrast, because attributes VIII–X (ecosystem function and scale-dependent features) are rarely directly assessed by biologists, the evaluation of these attributes was accompanied by relatively high uncertainty. Even so, workgroup members strongly advocated retaining these attributes in the BCG because of the importance of this information in making restoration decisions. There was considerable discussion regarding to which axis, the biological or stress axis, the attributes for ecosystem connectance and spatial and temporal extent of detrimental effect should be assigned. As an interim measure, the workgroup members recommended including these attributes as components of the biological axis primarily because of the importance state biologists placed on this information in predicting restoration or protection success. The BCG, thus, serves as a guide to interpret condition and is expected to be further refined as development and application continues.

The presence of non-native taxa in level 1 was also the subject of considerable discussion. Knowledge of the extensive occurrence of some non-native taxa in otherwise near-pristine systems conflicted with the desire by many to maintain a conceptually pure and natural level. Further discussion resulted in agreement that the presence of non-native taxa in level 1 is permissible only if they cause no displacement of native taxa, although the practical uncertainties of this provision were acknowledged. The resulting level descriptions, which allow for non-native species in the highest levels as long as there is no detrimental effect on the native populations, has practical management implications. For example, introduced European brown trout (*Salmo trutta*) have replaced native brook trout (*Salvelinus fontinalis*) in many eastern U.S. streams. In some catchments, brook trout only persist in stream reaches above waterfalls that are barriers to brown trout. The downstream reaches can be nearly pristine except for the presence of brown trout. In these places, if society decided to remove the introduced brown trout, and if stream habitat is preserved throughout the catchment, brook trout can potentially repopulate downstream reaches. In the use designation process, recognizing that the entire catchment has the *potential* to attain level 1 condition will inform the public that a very high quality resource exists.

Critical gaps in knowledge and scientific literature were uncovered during the development of the BCG. For example, the workgroup identified the need for regional evaluations of species tolerance to stressors. Tolerance information presented in the current version of the BCG tends to be based on generalized taxa responses to a non-specific stressor gradient. At that time, tolerance information was not available for most taxa and for many common stressors (temperature, nutrients, and sediments). In some cases, tolerance values are based on data collected in other geographic regions or for other purposes (e.g., van Dam's European diatom tolerances are used for North American taxa) (van Dam et al. 1994). In the future, availability of improved tolerance value information can be used to refine the BCG and improve its precision (e.g., Cormier et al. 2013; Whittier and Van Sickle 2010).

Additionally, taxa that are considered tolerant to stressors in one region of the country may not be similarly classified in another region. For example, long-lived taxa have generally been characterized as sensitive to increasing pressure and tend to be replaced by short-lived taxa in stressed systems. As such, the presence of long-lived taxa in a water body has been used to indicate high quality conditions, whereas the predominance of short-lived taxa may indicate degradation. However, in streams in the arid western U.S., extreme changes in hydrology might define the natural regime and an opposite trend might be observed: short-lived taxa can dominate the biological community in natural settings. In these systems, a shift to long-lived taxa may be an indicator of altered, less variable flow regimes due to flow management.

When the expert workgroup was initially developing the conceptual BCG framework (2000–2004), attributes VIII–X were not routinely measured as part of a state biological monitoring and assessment program. However, the state scientists participating in the workgroup deemed these attributes as ecologically important because the extent of ecosystem alteration has important environmental implications in terms of an individual water body's vulnerability to further effects from stressors, as well as potential for mitigation (Davies and Jackson 2006). The state scientists explained that they informally estimated ecosystem function, connectance, and extent of detrimental effects using different surrogate measures (e.g., shift in functional feeding groups) and/or measures of watershed condition (e.g., presence and connection of wetlands and streams, intact forests). This information was used to inform decisions on recovery potential for a water body and prioritize actions to protect high quality conditions.

Additionally, attributes IX and X might play an important role in evaluating longer term impacts, restoration potential, and recoveries. For example, ecosystem connectivity is fundamental to the successful recruitment into and maintenance of organisms in any environment. A single impacted stream reach in an otherwise intact watershed has far more restoration potential than a similar site in a basin that has undergone extensive landscape alteration.

A critical gap that was not discussed in 2005, but is now an area of intensive work, is predicting the impacts of climate change on aquatic systems. Gaining an understanding of how the BCG attributes (I–X) will behave under future climate scenarios, and developing approaches and indicators to measure these impacts, will be important future work for improving the BCG.

2.5 Conclusion

The conceptual BCG framework is a tool to help state water quality management programs better describe their ALU goals and measure increments of change in biological condition along a full gradient of stress—and to use that information to interpret existing conditions, identify high quality waters, and track progress towards achieving desired improvements. The BCG provides a common interpretative framework to assist in comparability of results across jurisdictional (e.g., county, state, national) and program (e.g., water quality and natural resource agencies) boundaries and to communicate this information to the public. In order to use the BCG, states will need to calibrate it to their own habitats and monitoring data and develop a numeric model. Although the BCG is a universal conceptual framework, quantitative calibrations are regionally data set-specific. Additionally, as an added benefit, state water quality management programs have reported that using expert consensus in developing BCGs has proven to be a valuable training tool for their technical staff and field crews. The panel interactions and development of consensus in interpreting data directly contribute to a more uniform approach and shared understanding of the aquatic ecosystems for which the state is responsible. Chapters 3 and 4 describe how a quantitative BCG model can be developed using expert panels and different approaches for quantification of the conceptual framework.

Chapter 3. Calibration of Biological Condition Gradient Models

The purpose of calibrating the BCG is to populate the conceptual model with quantitative data, develop quantitative decision rules to assign sites to BCG levels, and build a bridge from that model to management goals and endpoints. A calibrated BCG has both a narrative and a quantitative scientific description applicable to specific ecological regions or subregions. The BCG level descriptions can be used to describe the biological conditions associated with specific management goals and to support biological criteria development. The scientific description of the BCG can help make the management goals transparent to both decision makers and stakeholders. It can be used to assess baseline conditions and track incremental changes in condition.

This chapter proposes an approach to develop detailed narrative descriptions of BCG levels and attributes. Description and calibration of the BCG are achieved through consensus of expert opinion (Figure 7). The experts define the attributes, and the changes in those attributes, that characterize BCG levels and signal shifts to a different level. The outcome is a multiple attribute decision model that simulates the consensus expert decisions based on a set of quantitative rules. The next chapter provides three approaches to quantify the narrative BCG and develop numeric thresholds for site assignments.

Figure 7. Benthic macroinvertebrate and fish experts developing decision rules for freshwater streams in Alabama.

Use of professional expert consensus has a long pedigree in the medical field, including the National Institutes of Health (NIH) Consensus Development Conferences to recommend best practices for diagnosis and treatment of diseases.[5] In addition to the NIH consensus conferences, other researchers, institutes, and countries develop medical consensus statements, using both the NIH methods and others (Nair et al. 2011).

Recent environmental assessments developed using professional judgment have shown that experts are highly concordant in their ratings of sites, including marine benthic invertebrate communities in California bays (Weisberg et al. 2008). Another example is in nearshore marine environments assessed by an international panel covering European Atlantic, Mediterranean, American Atlantic, and American

[5] The program ran from 1977 to 2013. For more information, see: http://consensus.nih.gov/. Accessed February 2016.

Pacific habitats and experts (Teixeira et al. 2010). The approach has also been demonstrated effective for developing assessments of sediment quality (Bay et al. 2007; Bay and Weisberg 2010) and a decision model for fecal contamination of beaches (Cao et al. 2013). Likewise, in BCG development, aquatic biologists have come to very tight consensus on the descriptions of individual levels of the BCG, as well as very close agreement on the BCG level assigned to individual sites (e.g., Danielson et al. 2012; Davies and Jackson 2006; Gerritsen and Jessup 2007a; Gerritsen and Leppo 2005; Gerritsen and Stamp 2012; Gerritsen et al. 2013; Jessup and Gerritsen 2014; Kashuba et al. 2012; Snook et al. 2007).

All scientific and technical products, including biological indices used for assessment, include results of professional judgment and assumptions throughout (Scardi et al. 2008; Steedman 1994). The BCG expert consensus approach asks the experts to make judgments on the biological significance of changes in the attributes identified in Chapter 2. For this approach to be credible and valid, the panel should be comprised of experts with a wide and deep breadth of knowledge and expertise and not be constrained to a single agency in order to minimize internal bias. Additionally, it is essential that the expert logic in developing the decisions be fully documented so the rules will be transparent and understandable to those that were not engaged in the expert panel. The objective is to develop a set of decision rules that can be implemented by others not engaged in the expert panel.

3.1 Overview

The first step in calibration of the BCG is to develop detailed narrative descriptions of BCG levels and attributes specific to the water body type and region. Experts are given assemblage species composition and abundance data sets from the region for which they are developing the BCG. In order to minimize pre-conceived judgments, they are also given physical information about the sites (e.g., catchment area, slope, elevation, ecoregion, habitat type) but not the precise locations, land uses, sources, and stressor information. Following discussion of the conceptual model of the BCG, including detailed presentation on the description of the BCG levels and attributes, the experts are asked to put each sample site into one of the BCG levels. Each sample is discussed by the group, and facilitators elicit the reasoning used by the experts in their ratings. The median of the expert ratings is taken as the final BCG level for a sample (Gerritsen and Jessup 2007; Gerritsen and Leppo 2005; Gerritsen and Stamp 2012; Gerritsen et al. 2013; Jessup and Gerritsen 2014; Snook et al. 2007).

After an initial rating of at least 30 samples, the experts are asked to begin to articulate rules or guidelines that they use to make their decisions, starting with the highest level (BCG level 1) and working through level 6. Data evaluations and site assignments continue as rules are articulated and then revisited and further tested. In some situations, it may be necessary for the experts to use historical data and information to develop rules for the highest levels of the BCG when there are no or few samples in the data set that are representative of undisturbed or minimally disturbed conditions. Following the expert meetings, organizers and analysts examine the distributions of the quantitative data with respect to the initial proposed guidelines stated by the experts and the experts' actual BCG decisions. The distribution analysis forms the basis of quantitative boundaries around the experts' proposed rules, and analysts in turn develop quantitative rule-based models. Quantitative rules and performance are in turn reviewed by the expert panels to adjust rules or thresholds as necessary. Reviews and iterative recalibration are typically carried out by webinar and conference call. The panels also rate an independent set of test samples that were excluded from the calibration process.

The outcome of a full BCG calibration, including development of a quantitative model, includes:

- A current state-of-knowledge description of the biological assemblage of water bodies under pre-development, undisturbed condition to serve as a fixed, historic baseline (the level 1 prototype). If there are no BCG level 1 sites available, then this description may be based on historical observations, records, and/or data.
- Descriptions of each identified level of the BCG.
- A set of transparent rules for assigning sample sites to levels of the BCG.
- A quantitative model of the rules, or other technical approach, to assign new samples to levels of the BCG, without reconvening an expert panel.
- A set of BCG condition levels that can serve as management goals for classes of water bodies and as thresholds for biological criteria, if the state so chooses.

There are several key steps to the calibration process (Figure 8):

- **Assemble and organize data—**The BCG is developed using information and data from the state's existing biological monitoring program and/or other data sources (e.g., different data sets or regional pooled data from other states and federal agencies). The data should cover the entire range of conditions and stress within at least one ecological region. The data set should be sufficiently large with a well-defined approach for classification, identification of natural conditions, and criteria for reference site selection. Usually, the BCG cannot be calibrated within small jurisdictions or within urban or agricultural regions only—it requires data from outside the jurisdiction to ensure that the least stressed reference, as well as the full range of other stressors, are represented.
- **Conduct preliminary data analysis/data preparation**—Prior to the calibration workshop, the data must be put in a format that can be readily used by workshop participants. In addition, stressor-response relationships are examined to describe the responses of the assemblages and of individual taxa to the stress gradients represented in the data.
- **Convene expert panel**—The key component of BCG calibration is expert consensus of aquatic biologists on qualitative and quantitative descriptions of the BCG levels. Experts selected should be familiar with the water bodies, identities of species, and species and assemblage responses to stress in the regions of concern. The panel should include experts from not only the state biological assessment program but other state and federal natural resource agencies and research scientists from the academic community. Additionally, experts who regularly work with the regulated community can offer a level of assurance and interpretive assistance about the purpose and value of using the BCG in water quality assessments.
- **Develop quantitative BCG model**—Following the development of decision rules, one of several approaches can be applied to automate assigning water bodies to condition levels in the state database. Approaches discussed in Chapter 4 of this document include multiple attribute decision models, multivariate discriminant models, and development of thresholds for commonly used biological indices (e.g., multimetric indices (MMIs) or predictive model indices (e.g., observed over expected taxa [O/E])).
- **Test models, adjust, and recalibrate**—The development process is iterative and may require several passes through the process to converge on a consistent, locally calibrated BCG that is scientifically defensible.

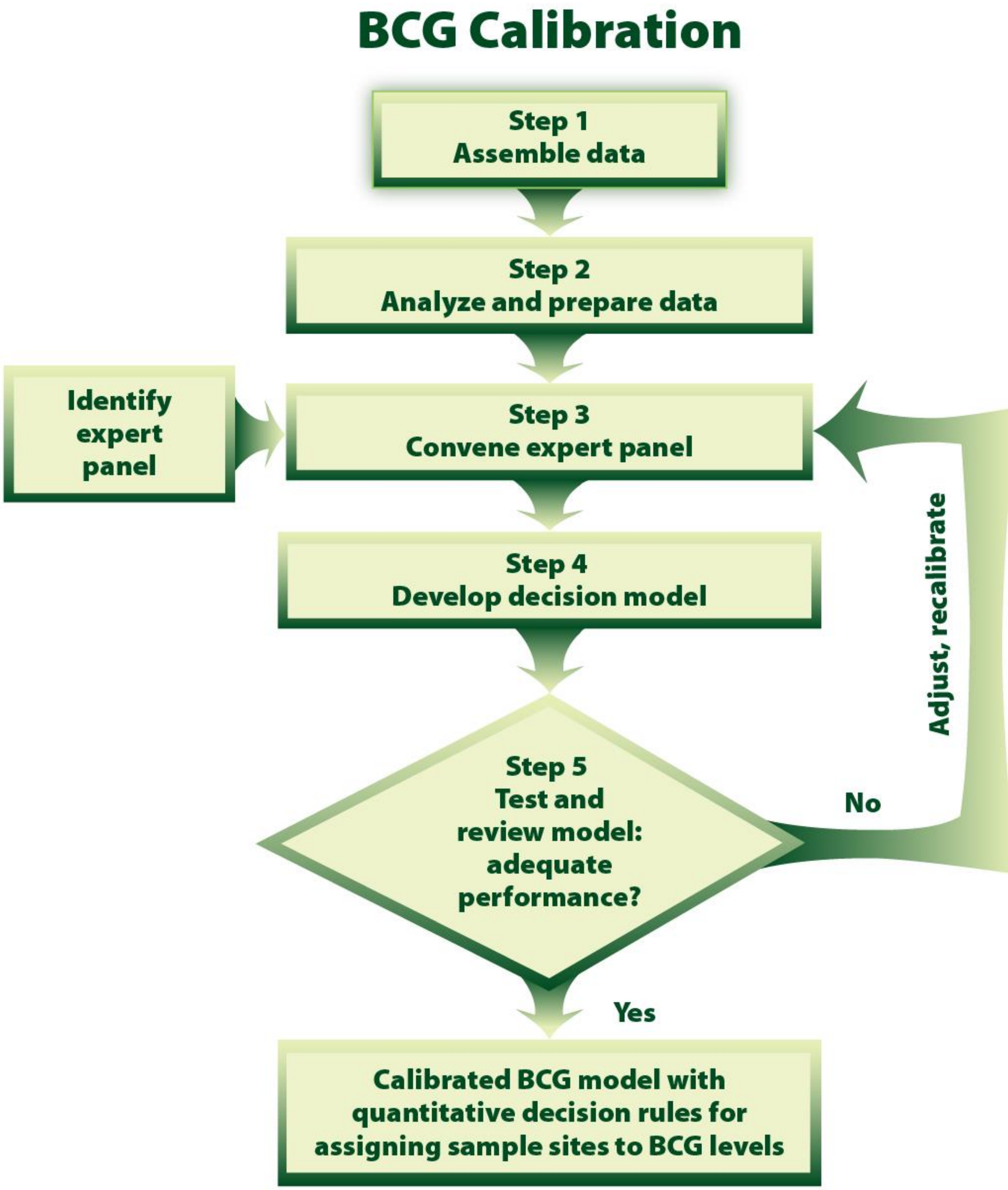

Figure 8. Steps in a BCG calibration.

3.1.1 Case Studies and Applications

Since 2005, several states or other entities (e.g., river basin associations, counties) have either calibrated, or are in the process of calibrating, the BCG (Table 4). Most of the BCG models that have been calibrated to date apply to perennial streams that are exposed to increases in temperature, nutrients, toxic substances, and fine sediments. This is the stream-type and stressor regime originally described by the conceptual model. Nevertheless, the model has been extended and calibrated to large rivers (Appendices B1 and B4; Bradley et al. 2014), estuaries (Appendix B2), coral reefs (Appendix B3; Shumchenia et al. 2015), and lakes (Gerritsen and Stamp 2014). Refinement of the model attributes to accommodate regional and water body type differences for water bodies other than streams and wadeable rivers has occurred without loss of model integrity. Thus, the BCG can be applicable to other aquatic ecosystems and stressors with appropriate modifications.

Section 3.2 below provides a detailed description of the step-by-step process that has been used to calibrate BCG models. Chapter 4 provides approaches to quantify the expert-derived BCG model, and case studies drawn from Table 4 illustrate different components of the process.

Table 4. BCG calibration and testing projects

State/Region	Water body type (if applicable)	Biological Assemblages	Objective	Status (Citation)
Alabama	Highland streams and wadeable rivers	Benthic macroinvertebrates and fish in high gradient streams	Calibrated BCG model and automated decision model for invertebrates (all streams) and fish (3 regions)	Complete (Jessup and Gerritsen 2014)
	Coastal plains streams	Benthic macroinvertebrates and fish in low gradient streams	Calibrated BCG model and automated decision model for invertebrates and fish	In progress
California	Streams	Algae	Calibrated BCG model and decision model for stream algae	In progress
Connecticut	High gradient streams and wadeable rivers	Benthic macroinvertebrates	Calibrated BCG model and automated decision model; also calibrated to Connecticut's macroinvertebrate MMI	Complete. (Gerritsen and Jessup 2007b)
		Fish	Calibrated BCG model and automated decision model; also calibrated to Connecticut's fish MMI	Complete (Stamp and Gerritsen 2011)
Illinois	Streams	Benthic macroinvertebrates and fish	Calibrated BCG model and automated decision model	In progress
Indiana	Streams and rivers	Fish	Calibrated BCG model and automated decision model	In progress
Maine	Streams and wadeable rivers	Algae	Calibrated BCG model to assign ALUs per Maine's 3 designated use classes and technical approach for benthic macroinvertebrates	Complete (Davies and Tsomides 2002; Davies et al. In press; Danielson et al. 2012)
	Wetlands	Benthic macroinvertebrates	Calibrating automated decision model to assess tiered designated ALU classes	In progress
Maryland, Montgomery County	Streams	Benthic macroinvertebrates and fish (quantitative), salamanders (qualitative)	Calibrated BCG model to communicate monitoring information on condition	Stamp et al. 2014

State/Region	Water body type (if applicable)	Biological Assemblages	Objective	Status (Citation)
Minnesota	Streams and wadeable rivers	Benthic macroinvertebrates and fish	Calibrated BCG model and automated decision model for nine stream types; also incorporates Region 5 coldwater results	Complete (Gerritsen et al. 2012)
	Lakes	Fish	Calibrated BCG model and automated decision model for four lake types	Complete (Gerritsen and Stamp 2014)
New England	High gradient streams and wadeable rivers	Benthic macroinvertebrates	Cross-calibrated BCG model and automated decision model for multiple sampling methodologies	Complete (Snook et al. 2007)
New England	Large rivers	Fish	Calibrated BCG model and automated decision model	In progress
New Jersey	High and low gradient streams and wadeable rivers	Benthic macroinvertebrates	Calibrated BCG model and automated decision model	Complete (Gerritsen and Leppo 2005)
	Streams and wadeable rivers	Diatoms	Calibrated BCG model and automated decision model	In progress
Pennsylvania	High gradient streams and wadeable rivers	Benthic macroinvertebrates	Conceptual model and verbal description of BCG levels, calibrated to Pennsylvania's MMI	Complete (Gerritsen and Jessup 2007a)
Puerto Rico and U.S. Virgin Islands	Stony coral reefs	Stony corals and resident reef fish	Calibrated BCG model and automated decision model	In progress (Bradley et al. 2014)
Rhode Island	Estuaries	Seagrass extent, benthic community, shellfish production, primary productivity in Greenwich Bay, Rhode Island	Conceptual BCG model anchored in natural conditions prior to 1850 and showing changes	Complete (Shumchenia et al. 2015)
		Habitat mosaic indicator as measure of whole system condition for Narragansett Bay	In progress	In progress preparation, Shumchenia et al. in review)
Upper Mississippi River Basin	Large rivers	Fish	Calibrated BCG model and automated decision model	In progress
Vermont	Streams and wadeable rivers	Benthic macroinvertebrates	Calibrated BCG model and biological criteria	VT DEC 2004

3.2 Step One: Assemble and Organize Data

Evaluating data quality and preparing it for rule development is critical for an efficient and effective expert panel meeting. The data should cover the entire range of conditions and stress within at least one ecological region. Typically state databases have been used in the stream BCGs developed to date, but large regional databases, either a single data set or pooled data sets, have also been used (e.g., Upper Mississippi Basin BCG and New England River BCG (see Appendices B1 and B4)). Combining data sets presents a unique set of challenges for experts in interpreting site data and detecting consistent patterns of biological change in response to increasing stress. If different data sets are combined, decisions on how the data sets are reconciled must be well documented for the experts to successfully use the data in rule development. When BCG rules are developed for more than one assemblage, typically different data sets are used for each assemblage and the rules are developed and applied independently. The rules for the different assemblages are tested jointly as a later step in the BCG model development.

There are three tasks required for assembling and organizing data prior to convening an expert panel:

1. **Obtain Data**—In preparation for the calibration process, relevant data are extracted from the database. Data should include the biological survey (taxonomic identification and counts) and all related data on the geo-referenced sampling site: locations and characteristics; catchment data including area, slope, land use characteristics, and physical habitat; chemical water quality data; and field observations by sampling personnel. Evaluation and documentation of the quality of the data set is an essential component of the BCG approach, including documentation of technical issues and concerns that should be further addressed through additional data collection and analysis. Section 3.2.1 discusses elements of a data set and monitoring program that should be evaluated and documented.
2. **Determine Natural Classification**—In order to prevent natural variability from confounding responses to stress, it is necessary to determine a natural classification system for the water bodies under consideration (if not already complete) (USEPA 2013a). If there is only a collection of data, and no agreed-upon classification system, substantial analytical effort might be needed to develop it. Classification is beyond the scope of this document; see Barbour et al. (1999), Hawkins et al. (2000a), USEPA (2013a), Olivero and Anderson (2008), and Olivero Sheldon et al. (2015) for references to classification approaches for freshwater streams. Selection of a classification method was one of the first tasks the coral reef expert panel undertook prior to successful rule development (see Appendix B3). The classification decision has implications for statistical sampling design and monitoring protocols.
3. **Organize Data Tables**—A comprehensive and relational database is a requirement for a high quality monitoring program (USEPA 2013a). Data can be organized in spreadsheets for the panel workshops (see Figure 12 for an example of datasheet used in BCG development to date). For permanent storage, retrieval, archiving, and to maintain a quality record, a relational database will be necessary (e.g., Oracle®, MS-Access®, Sequel Server®).

Quantitative assessment within the BCG framework requires consistent, high quality biological, physical, chemical, and geographic monitoring information. The technical foundation of monitoring determines the degree of confidence with which the information can be used to support water quality management decision making, including calibration of the BCG. This section describes data requirements consistent with EPA's recommended program review of biological assessment programs (USEPA 2013a).

All BCG developments to date have used existing state or federal agency monitoring data. There have been no monitoring programs specifically designed for BCG development. However, recommendations on the technical elements of a monitoring program that would produce good data for BCG development are not different from the requirements for a high-quality program specified by EPA (2013a) and are discussed below. This document is not guidance for monitoring design, optimal effort, or sampling methods. Instead, it focuses on minimum requirements for BCG development, including consistently sampled aquatic biota; water quality and habitat observations adequately matching the biological sampling; and land use/land cover information (e.g., from the NHD coverage). Consistency and adequacy of a data set are evaluated by the expert panel, analysts, and facilitators, and documentation of BCG development includes recommendations on specific technical areas where further development would strengthen or refine the quantitative BCG model and underlying decision rules.

3.2.1 Data Requirements: Understanding the Quality of the Data Set

EPA described 13 technical elements contributing to quality of biological assessment programs (USEPA 2013a). These elements are listed below and constitute the technical underpinnings important for a biological assessment program to be able to discriminate levels of condition along a gradient of disturbance (Table 5). Selected elements of biological assessment program design and data collection, compilation, and interpretation important for BCG development are discussed below. For a more complete description, see EPA's *Biological Assessment Program Review: Assessing Level of Technical Rigor to Support Water Quality Management* (USEPA 2013a). It is recommended that these elements be considered when assembling a data set for BCG calibration. Understanding the technical strengths and limitations of the data sets to be used in the calibration will help guide development of the BCG and its application.

Table 5. Definitions of the technical elements (USEPA 2013a)

	Technical Element	**Definition**
Biological Assessment Design	Index Period	A consistent time frame for sampling the assemblage to characterize and account for temporal variability.
	Spatial Sampling Design	Representativeness of the spatial array of sampling sites to support statistically valid inference of information over larger areas (e.g., watersheds, river and stream segments, geographic region) and for supporting WQS and multiple programs.
	Natural Variability	Characterizing and accounting for variation in biological assemblages in response to natural factors.
	Reference Site Selection	Abiotic factors to select sites that are least impacted, or ideally, minimally affected by anthropogenic stressors.
	Reference Conditions	Characterization of benchmark conditions among reference sites, to which test sites are compared.
Data Collection and Compilation	Taxa and Taxonomic Resolution	Type and number of assemblages assessed and resolution (e.g., family, genus, or species) to which organisms are identified.
	Sample Collection	Protocols used to collect representative samples in a water body including procedures used to collect and preserve the samples (e.g., equipment, effort).
	Sample Processing	Methods used to identify and count the organisms collected from a water body, including the specific protocols used to identify organisms and subsample, the training of personnel who count and identify the organisms, and the methods used to perform quality assurance/quality control checks of the data.
	Data Management	Systems used by a monitoring program to store, access, and analyze collected data.
Analysis and Interpretation	Ecological Attributes	Measurable attributes of a biological community representative of biological integrity and that provide the basis for developing biological indices.
	Discriminatory Capacity	Capability of the biological indices to distinguish different increments, or levels, of biological condition along a gradient of increasing stress.
	Stressor Association	Relationship between measures of stressors, sources, and biological assemblage response sufficient to support causal analysis and to develop quantitative stressor-response relationships.
	Professional Review	Level to which agency data, methods, and procedures are reviewed by others.

3.2.1.1 Biological Assessment Design Elements

The first four technical elements are particularly critical aspects of sampling design to consider when evaluating data for BCG calibration, and they involve selection of sites and times for sampling to obtain representative and statistically valid information (USEPA 2013a). The fifth element, reference condition, is also discussed below but in relation to its role relative to the BCG benchmark, BCG level 1 (e.g., anthropogenically undisturbed reference condition).

Index Period

Sampling index periods are selected based on known ecology to minimize or account for natural variability, maximize sampling gear efficiency, and maximize the information gained on the assemblage (Barbour et al. 1999; USEPA 2013a). For temperate fresh water bodies, index periods are typically a span of 3–6 months during the growing season.

Spatial Sampling Design: Representative of Stress Gradient

The objective of BCG calibration is to characterize the biological response across a generalized stress gradient from undisturbed to highly disturbed conditions. The BCG should be developed for specific natural classes, such as ecoregions or physiographic provinces. Sample coverage must be representative of the ecoregion(s), as well as the stress gradients that can occur. Case examples of characterizing stress gradients are given later in this chapter (section 3.3.1) and discussed more generally in Chapter 5. Achieving representativeness might require using data from outside of the jurisdictional boundaries of a state so that ecoregional expectations are as fully sampled and defined as possible. In addition to representativeness, the data should have sufficient sample size to support the calibration. As a rule-of-thumb, 30 or more samples for each water body class (at a minimum for levels 2–5, and levels 1 and 6, if regionally available) are generally required (see natural classification below). If samples are not sufficient, then BCG development should be delayed until enough data are acquired.

Calibration of the BCG model requires data points (samples) along the stress gradient from low to high levels of stress. An expert panel examines the sites and assigns the sites to BCG level based solely on the biological information. Having the stress information ensures that the expert panel sees sites that are representative of the stress gradient. Ideally, the data set needs to include the full gradient of conditions and complement of stressors (e.g., pollution sources, invasive species, habitat disturbances) that are common to a state or region, such that the full gradient of assemblage response is included in the model development.

Natural Variability: Classification

Biological assessment based on knowledge of the biota under undisturbed or minimally disturbed reference sites forms expectations for natural conditions. Many natural regional and habitat characteristics (e.g., stream size, slope, dominant natural substrate) also affect the species composition of undisturbed water bodies. Accordingly, a critical step in developing a biological assessment program is to classify the natural conditions to the extent that they affect the biological indicators (e.g., Barbour et al. 1999; Gibson et al. 1996; Hawkins et al. 2000a). The term classification includes development of continuous models that explain natural variability of biological assemblages. For example, fish species richness is strongly dependent on catchment area or average flow. Modeling approaches that combine both discrete and continuous variables (e.g., general linear models) may be especially powerful if the data support them. Failure to properly classify sites can cause the BCG calibration to fail, yielding assessment errors that can undermine confidence in results. Classification of natural conditions should be complete and satisfactory to experts. If not, time and resources will be necessary to develop the classification system.

Reference Site Selection

Obtaining a representative stress gradient requires that the data set is large enough to include the full range of disturbance, from undisturbed to highly disturbed conditions. Data owners and field personnel should document the comprehensiveness of the data set with respect to coverage of the full range, or not, of disturbance. It might be necessary to obtain data from neighboring states or regions to ensure that the gradient is represented in the data set. A minimum of 30 to 40 sites might be sufficient to calibrate the BCG depending upon both the characteristics of the natural system and the quality of the data set. Characterization of the quality of reference sites is essential to defining the range of conditions in the data set the experts will be using to develop decision rules. The criteria used by states to select their reference sites inform this determination.

Reference Condition

In this document, the terms "undisturbed," "minimally disturbed," and "least disturbed" conditions are used when referring to the level of anthropogenic stress to which a water body and its surrounding watershed may be subject. These terms are well defined by Stoddard et al. (2006). The level of stress associated with the reference sites used by the state to define reference condition is the critical information needed for BCG calibration. In many cases, the state's reference condition is not comparable to the BCG benchmark for undisturbed or minimally disturbed conditions. This is important information, not only for the BCG calibration but also for water quality program managers and the public.

BCG calibration is not based on least disturbed reference sites, because least disturbed sites are typically the "best of what is left" and may mistakenly be perceived by the public as the best that can be because undisturbed conditions no longer exist (e.g., Dayton et al. 1998; Papworth et al. 2008; Pauly 1995). In this case, expectations for improvements might be set lower than the potential for a water body to improve. Part of the BCG process can include developing a description of undisturbed conditions that may include consideration of contemporary, empirically least stressed sites and historical descriptions; paleolimnological investigations; and museum records. The description of an undisturbed condition may be narrative and perhaps incomplete, but its documentation helps provide a transparent and clear framework for the public to understand what biologically may have already been lost from their waters as well as potential for what could be restored. In many of the BCGs that have been developed, undisturbed and minimally disturbed conditions have been combined for practical purposes and categorized as representing BCG levels 1 and 2.

3.2.1.2 Methodological Elements

The second set of technical elements are aspects of quality in sampling, processing, and data management. Data used for calibration must be consistent, or be made consistent in post-processing. It is especially important to examine methods when biological assessment data from multiple sources are to be pooled. For information on combining data derived from multiple sources, see Gerritsen et al. (2015). Elements of sampling methodology include:

Taxa and Taxonomic Resolution

The biological response data should be the taxonomic composition and related information from one or more biological assemblages in water bodies: benthic macroinvertebrates, fish, periphyton, aquatic macrophytes, phytoplankton (lakes and estuaries), and zooplankton (lakes and estuaries). To develop the model, a knowledgeable panel of experts is required for each assemblage.

Experience has shown that "lowest practical" identification, to species when possible, is superior for BCG calibration, because species differ in their characteristics within genera. Species identification is necessary for fish assemblages, but genus-level identifications are adequate for BCG calibration using benthic macroinvertebrates. When pooling data, the taxonomic resolution must be standardized to the lowest common level among the data sets.

Sample Collection

Field methods should be consistent and well-documented. The objective of the sampling methods should be to obtain consistent samples that are representative of the target biological assemblage (see Barbour et al. (1999) and USEPA (2013a) for discussions of sample collection methods). The BCG has been cross-calibrated for several sampling methods used in New England and in the Upper Midwest (Gerritsen and Stamp 2012; Snook et al. 2007). Where possible, initial BCG development in a new region is done with data from a single sampling methodology and then can be calibrated and tested with data generated using different sampling methods. Level of effort is a key consideration in sample design. Many of the BCG attributes (attributes I, II, VI, VII, IX, and X) are particularly sensitive to level of effort. Certain key taxa may be sparse, seasonal, or patchy in their distribution and easily missed by a standardized field collection method. In making a site assessment, other supplemental information (e.g., natural history surveys, fishery agency reports and observations, academic studies), beyond just the collected samples, should be included in making a level determination. This will lend an additional layer of confidence and improve the result.

Sample Processing

Laboratory processing of samples (except fish) is recommended (USEPA 2013a; Yoder and Barbour 2009). Macroinvertebrate and diatom samples are typically processed to a standardized count representing a constant sampling effort. In some cases, if subsampling efforts are mixed, it is possible to randomly subsample larger efforts to smaller efforts (e.g., 300-count subsamples randomly subsampled further to match 100-count subsamples).

Data Management

Identification of reference and stressed sites requires that the monitoring database be comprehensive, including watershed and site characteristics, habitat measurement, and physical and chemical water quality measurements. Physical and chemical measurements should be made at the same time and place that the biological community information is collected. Non-biological data, including catchment area, slope, land use, site, habitat, and physical and chemical water quality data are used to determine a site's natural classification and stressor status, such as whether it is a reference site or a stressed site, and where it is located along the stress gradient.

Data should be stored in a relational database so that queries can retrieve relevant information (e.g., biological data, chemical data, physical measurements) on site, geo-referenced location, multiple measurements from multiple sampling times, and catchment data. Data stored in spreadsheets or warehoused in such a way that physical, chemical, and accurate geo-reference cannot be located are of limited use and might require substantial costs to fill in missing information and for quality assurance (Gerritsen et al. 2015). Exceptions should be made for historic information and data that may not be amenable to spreadsheets. These data may not be suitable for a relational database but should be retained because they may provide important qualitative information and context that can be used to inform BCG development.

3.2.1.3 Analysis and Interpretation Elements

Ecological Attributes

These are the measurable attributes of a biological community that are representative of biological integrity and which provide the basis for developing a BCG model. The BCG attributes (Table 1) are the basis for this technical element. The selection of attributes might depend on the spatial scale and specific water body being assessed. Each attribute provides some information about the biological condition of a water body. Combined into a conceptual model comparable to the BCG, the attributes can offer a more complete picture about current water body conditions and also provide a basis for comparison with naturally expected water body conditions. All states that have applied a BCG for streams, rivers, and wetlands have used the first seven attributes that describe the composition and structure of the biotic community on the basis of the tolerance of species to stressors. Where available, they have included information on the presence or absence of native and nonnative species, and, for fish and amphibians, used measures of overall condition (e.g., size, weight, abnormalities, and tumors). Though not measured directly in state or tribal stream biological assessment programs, the last three BCG attributes of ecosystem function and connectedness and spatial and temporal extent of stressors can provide valuable information when evaluating the potential for a stream, river, or wetland to be protected or restored.

Discriminatory Capacity

This technical element addresses the degree of sensitivity of the BCG model in distinguishing incremental change along a continuous gradient of stress. Detailed descriptions of biological change along a gradient of stress can provide the conceptual basis for refined ALUs for specific ecotypes and regions leading to biological criteria development. Additionally, depending on the sensitivity, or discriminatory capacity, of the BCG model, the information can be used to help identify high quality waters and establish incremental goals for improving degraded waters. Six general increments of change can be described for each of the BCG's ecological attributes (for example, see Table 3). These incremental changes can serve as a template for developing biological criteria that represent aspects of biological integrity and which show a predictable, measurable response to increasing levels of stress.

The number of increments that can realistically be distinguished in a BCG model is dependent not only on the water body ecotype and natural classification factors that define biological assemblage characteristics, but also on the effect of anthropogenic stressors. For example, the sensitivity of an index developed for a forested, high-gradient stream might support distinguishing five or even six increments of quality along a continuous stressor gradient, while an intermittent, seasonal, or desert stream may yield fewer increments. Some of this difference is due to inherent natural characteristics of the assemblages, and some might be due to current limitations of science and practice.

Stressor Associations

Stressor association refers to the use of biological assessment data at appropriate levels of taxonomy to develop relationships between measures of biological response and anthropogenic stressors, including both stressors and their sources (Huff et al. 2006; Miller et al. 2012; Yuan 2010; Yuan and Norton 2003). This element includes examination of biological assessment data for patterns of response to categorical stressors (Riva-Murray et al. 2002; Yoder and DeShon 2003; Yoder and Rankin 1995a). A capability for developing these relationships extends the use of biological assessments from assessing condition to informing identification of possible causes and sources of a biological impairment at multiple scales.

Stressor association is directly dependent on a high level of technical development of other elements, particularly the elements for spatial sampling design, taxa and level of taxonomic resolution, database management, and discriminatory capacity. These elements are important building blocks for the data collection and analysis needed to more confidently identify stressors and their sources and to estimate stressor-response relationships. For example, the ability to estimate these relationships relies on paired stressor and response sampling at appropriate spatial and temporal scales and a level of taxonomic resolution and index sensitivity sufficient to detect incremental biological changes along a stress gradient. Also, a relational database that supports complex queries enables efficient and full utilization of data. A high level of technical development for each of these elements and others provides the foundation for stressor association.

Professional Review

Professional review and testing of the BCG quantitative decision rules should be conducted by experts outside of the panel to evaluate and improve model objectivity and scientific defensibility and to refine and improve the model. Review by outside experts can be used to refine and improve the model. Because of the specific knowledge of the expert panel for any given BCG, discussion with the outside peer reviewers is essential. Technical expert review across expert panels has not yet been conducted for the BCGs developed to date, but it is planned as a pilot.

3.3 Step Two: Preliminary Data Analysis and Data Preparation

Before an expert panel is convened to describe the BCG levels, it is necessary to reduce and prepare the data for the panel's use during the workshops and webinars. In addition, it is useful to conduct exploratory data analyses to visualize empirical relationships of the biotic assemblages. Analysis and data preparation include:

1. **Characterizing Stress Gradients—**Identifying stress gradients to select sites for BCG calibration that are representative of stress gradients in the region, from undisturbed to highly disturbed levels of condition. If undisturbed conditions do not exist, the level of disturbance should be recorded and efforts made to collect historical data and records that may help the panel develop a conceptual, descriptive condition level absent of anthropogenic influence (BCG level 1). See Chapter 5 for further discussion.
2. **Analyzing Taxon Response Relationships—**Using the stress gradient(s) to examine stressor-response of individual taxa to augment known or surmised species tolerances and traits with empirical information on responses observed in the field, as well as to develop species distribution maps of species observed in the data set. This step also ensures that all panelists have the same information available to them, as some panelists may be more familiar with the monitoring data set than others. See Chapter 5 for further discussion.
3. **Preparing Data Work Sheets—**Identifying and formatting a calibration data set for the workgroup's calibration exercise.

This section discusses the preliminary analysis and data manipulation prior to calibration workshops.

3.3.1 Data Preparation: Characterize Stress Gradients

Water bodies are subject to a wide variety of anthropogenic stressors, and multiple stressor situations are common. However, few state data sets are sufficiently large and complete to be able to analytically separate the effects of individual stressors. To help select sites for the calibration exercises, a practical approach is to consider all stressors together without regard for interactions among them (Smith et al. 2001), or to use aggregate land cover as a summation of sources of potential stressors to streams (e.g., Landscape Development Intensity Index [LDI]; Brown and Vivas 2005). Stressor-response analysis with multiple, independent stressor gradients is currently an area of active research (e.g., Baker and King 2010; Norton et al. 2015), but it is beyond the scope of this document.

<u>Quantitative Gradients</u>

Identifying stress gradients relevant to the data sets at hand will be facilitated by some exploratory data analysis to identify biological responses to the stressors. Scatter plots are generally the most useful and efficient, but more detailed analysis can be done if desired, including regression analyses, quantile regression, and classification and regression tree (CART) analysis (Death and Fabricius 2000), and other models. The purpose of these analyses is not diagnostic, as BCG calibration does not include identifying the most likely causes for biological impairment. The purpose is to develop a database suitable for discerning patterns of biological response to increasing levels of stress.

Scatter plots can be examined for every stressor that will be included in a stress gradient, as well as for aggregated sources of stress such as land use/land cover. For this purpose, scatter plots are simple graphical displays of a response variable on the y-axis, against a presumed correlated parameter on the x-axis (e.g., Figure 9). Examples of stress variables examined for some of the BCG applications are shown in Table 6.

Table 6. Examples of quantitative stressor variables that have been used for BCG projects

Project	Quantitative Disturbance gradient
Minnesota streams	Human Disturbance Score (HDS)
Connecticut streams (fish)	% Developed area
Minnesota lakes	% Urban + Agricultural + Mining land use
	Trophic State Index
Maine stream algae	Total phosphorus
Maine stream benthic macroinvertebrates	% Impervious surface
Northern Piedmont region of Maryland	% Impervious surface
	Habitat index
Alabama	Human Disturbance Gradient (HDG)
Illinois	Habitat index
	% Impervious surface
	Total nitrogen
Indiana	% Impervious surface

3.3.1.1 Example—Using Land Use/Land Cover Indicators to Develop a Quantitative Stress Gradient (Minnesota, Alabama, Maryland Piedmont)

Measures of land use and land cover have been used as surrogate indicators of stressor effects. Table 6 contains a list of these type of indicators that have been used for GSA development (For more information on the GSA, see Chapter 5). In the Northern Piedmont of Maryland, the workgroup selected imperviousness as a primary stress indicator (Stamp et al. 2014, see Chapter 6). The percent imperviousness in a watershed or a catchment was available for all sites in the data set. Based on scatterplots like the one shown in Figure 9, the level of imperviousness has a clear impact on biological assemblages. Imperviousness was considered during the sample selection process to ensure that the full stress gradient was represented in the BCG model calibration data set. Imperviousness was also used to generate the taxon-response plots that helped inform BCG attribute assignments (see section 3.3.2).

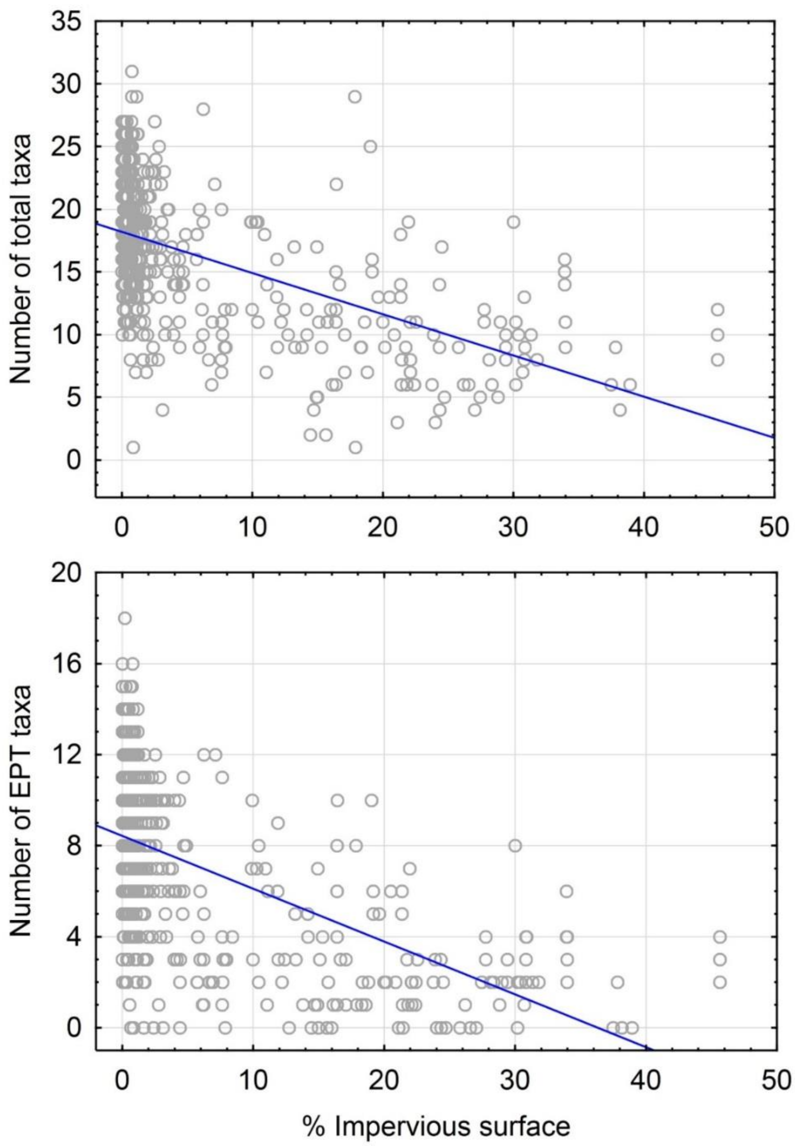

Figure 9. Scatterplots of number of total taxa (upper) and number of EPT taxa (lower) versus % impervious surface in the macroinvertebrate data set for streams in the Northern Piedmont of Maryland. Plots are fit with a linear trend line.

In some instances, quantitative stress gradients have been developed to capture multiple stressors in one integrated score. Examples include Minnesota's Human Disturbance Score (HDS) and Alabama's Human Disturbance Gradient (HDG). Input variables for the Minnesota HDS and the Alabama HDG are listed in Table 7 and Table 8, respectively. The Alabama HDG utilizes the LDI (developed by Brown and Vivas (2005)), which associates land uses with a scale of disturbance intensity and weights the index score based on land uses in the upstream catchment. Alabama Department of Environmental Management (ADEM) has used the HDG to assign its stream reaches to one of eight HDG categories based on the percentile of its overall HDG score, with categories 1–3 representing the top 25th percentile of watershed condition.

Table 7. Input variables for Minnesota Pollution Control Agency's (MPCA's) HDS (MPCA 2014a)

HDS Metric	Scale	Score
Animal unit density	watershed	10
Feedlot density	watershed	adjust
Feedlot proximity	local	adjust
Point source density	watershed	10
Point source proximity	local	adjust
Percent disturbed riparian habitat	watershed	10
Riparian condition rating	local	10
Percent agricultural land use	watershed	10
Percent agricultural land use within 100-m riparian buffer	watershed	adjust
Percent agricultural land use on ≥ 3% slope	watershed	adjust
Percent impervious surface	watershed	10
Urban land use proximity	local	adjust
Percent of stream distance modified by channelization	watershed	10
Site channelization rating	local	10
Road/stream intersection (road crossing) density	watershed	adjust

Table 8. Input variables for Alabama's HDG (Source: Lisa Huff, ADEM, personal communication)

The LDI associates land uses with a scale of disturbance intensity and weights the index score based on land uses in the upstream catchment, such that land uses that produce higher levels of disturbance receive higher LDI coefficients (Brown and Vivas 2005).

Variable	LDI coefficient	Source
Population density/km^2	1	2000 U.S. Census
% Urban	8	2006 National Land Cover Database
% Barren	8.6	
% Pasture	3.1	
% Cropland	4.7	
Road density	8.3	2010 Census TIGER/Line Shapefiles
# Stream/road crossings	8.3	

Ordinal Stress Gradient(s)

Where there are many measured stressors, it is possible to develop an ordinal, generalized stress gradient by summing and ranking the number of different stressors observed at distinct sites. A site is given a score of 1 for each stressor observed there (e.g., copper above a chronic screening threshold, excess nutrients, poor habitat score, upstream discharge), and the site score is the sum of all stressor scores. Sites with scores of 0 are candidates for "least stressed" within the context of the region. Categories of stress can be defined using measured stressors (e.g., contaminants and habitat condition) from the monitoring data, with watershed information that identifies sources of stress. The categories are a mixture of both sources and measured stressors and will inevitably be correlated to some extent. The categorization can identify a gradient of stress levels comparable to levels of disturbance (e.g., undisturbed, minimally disturbed, highly disturbed conditions (Stoddard et al. 2006)).

Stressors, whether individual or categories, can be screened by examining the response of individual taxa to the stressor or source (Figure 10)—if there is no response, the stressor should not be used in developing the ordinal gradient.

After relevant stressors and sources have been categorized, sites can be identified according to the number of stressors and sources in low, medium, or high categories. Sites where all stressors and sources are "low" qualify as least stressed, and sites where many stressors are "high" qualify as most stressed. Depending on the number of sites and stressors, intermediate categories can also be identified. Depending on the level of stressors detected in the "least stressed" category, undisturbed or minimally disturbed conditions may not be included in the data set. If this is the case, expert judgment on undisturbed or minimally disturbed conditions can be elicited based on historical observations, records, and data. Although a qualitative assessment, this information provides context for the quantitative information (e.g., "least stressed" conditions do not present undisturbed or minimally disturbed conditions).

3.3.1.1 Example: Connecticut Ordinal Stress Gradient

To identify sites to use in a BCG calibration exercise for Connecticut, analysts developed an ordinal stress gradient to apply to sample sites. The approach was to screen measured stressors for association with biological measurements and identify thresholds (stressor concentrations) below which no effects or association could be detected and screening thresholds above which association was strong. This was not an attempt to do a causal analysis (Norton et al. 2015), but simply a screening based on pairwise associations.

Connecticut DEP had sampled dissolved metals and several other water quality parameters simultaneously with each stream biological sample. For example, Figure 10 shows the number of Plecoptera (stonefly) taxa and dissolved copper concentration in Connecticut stream sites. High numbers of stonefly taxa (> 4) only occur when copper is less than 0.008 mg/L, and nearly all samples where copper was greater than 0.008 mg/L had fewer than 4 stonefly taxa (Figure 10). For the stressor gradient, the threshold for low copper stress was set at < 0.008 mg/L, and the threshold for high copper stress was set at > 0.008 mg/L. Note that there is not inference of causality, only screening of associations.

Stress categories were identified for Connecticut monitoring sites based on land use and water chemistry parameters in the database. Urban land use, natural land cover, population density, and chloride concentration were all good predictors of biological condition. Connecticut Department of Energy and Environmental Protection (CT DEEP) defined six stress categories for streams, based on the distribution of stressor parameters in the database.

CT DEEP's thresholds for the "least stressed" category (Table 9) were determined from stressor-response scatterplots of sensitive taxa in the samples versus the stressor parameters (dashed line in Figure 10). Screening thresholds for metals (Table 9) were determined from stress-response scatterplots of number of mayfly or stonefly taxa in the samples vs. metal concentrations (Figure 10). These two orders are generally considered highly sensitive to metal contamination (e.g., Buchwalter and Luoma 2005). Metals not included in Table 7 (aluminum, cadmium, mercury, lead, selenium) were either not associated with biological responses (no observable stress-response), or they were not detected in most observations in the data set. Using the criteria of Table 9, least disturbed sites were identified as sites with all eight stressor values in the "least stressed" category, and highly disturbed sites were identified as sites with four or more stress values in the "high" category. The screening allowed selection of sites for calibration to cover the range from "least disturbed" to putative "highly disturbed."

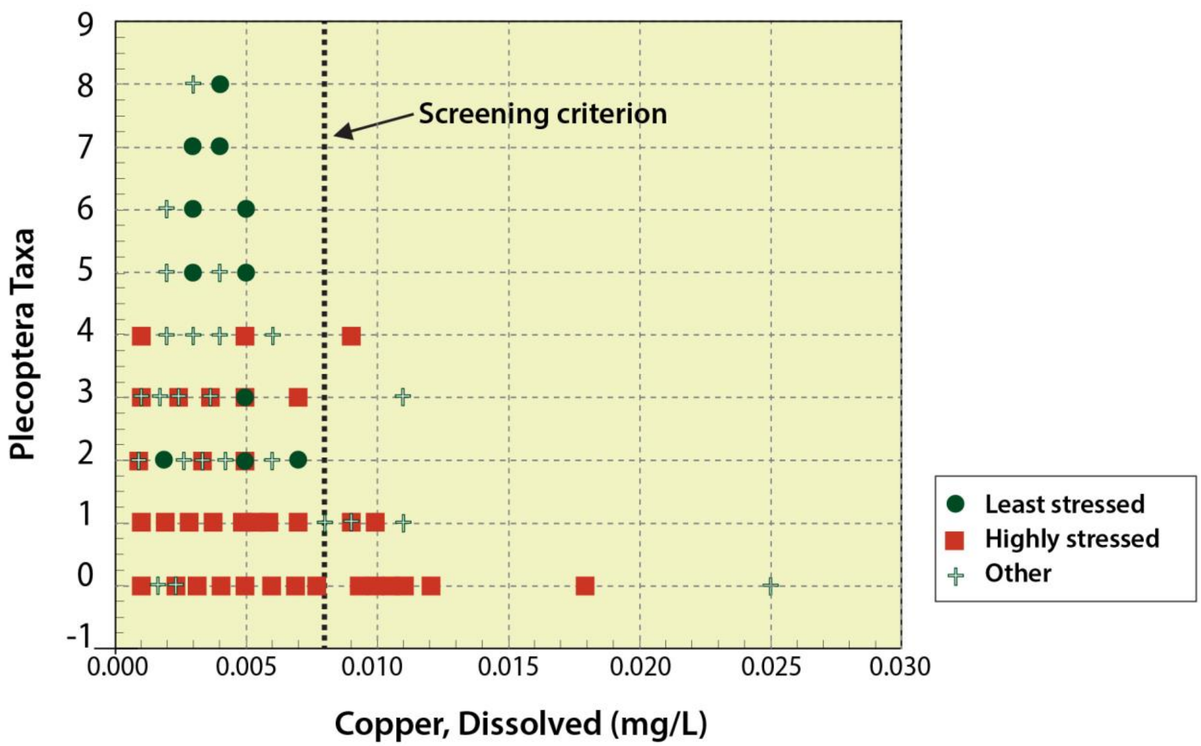

Figure 10. Number of Plecoptera (stonefly) taxa and dissolved copper (Cu) concentration, Connecticut sites. The screening criterion, (0.008 mg/L Cu) was estimated by eye from the presence of stoneflies at low Cu concentrations, and their near absence above 0.008 mg/L Cu. In the calibration, least stressed sites were required to have Cu < 0.008 mg/L (among other criteria). The screening criterion separates sites with no detectable influence of copper from those where copper may be a factor (among others) in loss of Plecoptera.

Table 9. Example screening thresholds for stressor gradient (Connecticut)

Parameter	Stress Category				
	Least Stress	Slight Stress	Moderate Stress	High Stress	Severe Stress
Catchment parameters					
Natural land cover*	> 80%	70%–80%	60%–70%	< 60%	
Developed land	< 10%		10%–25%	> 25%	
Impervious surface	< 4%		4%–10%	> 10%	
Water quality, non-metals					
Chloride	<15 mg/L	15–20 mg/L	20–30 mg/L	> 30 mg/L	
Water quality, metals					
Copper	< 0.008 mg/L			≥ 0.008 mg/L	
Iron	< 0.4 mg/L			≥ 0.4 mg/L	
Nickel	< 0.01 mg/L			≥ 0.01 mg/L	
Zinc	< 0.02 mg/L			≥ 0.02 mg/L	
Decision criteria for stress level	All parameters lowest stress category	Land cover or chloride Slight category; All others lowest category	Any one nonmetal allowed High category; All others Moderate or lower	Up to three non-metals High; Any metals High	All non-metals High-Severe; Any metals High

*defined as the sum of deciduous, conifer, open water, and all wetland categories

3.3.2 Data Preparation: Analyze Taxon Stressor-Response

An early task of the expert panel is to assign taxa to the attributes I through VI for development of stream and river BCGs. These are the primary attributes that are used to assess sites among BCG levels 2 through 6 for streams and rivers. Attribute VII, which provides information on organism condition (especially of long-lived organisms), is a general indicator of organism health, such as deformities, anomalies, lesions, tumors, or excess parasitism. This attribute has been used with great success in indices based on the fish assemblage. To date, attributes VIII through X have not been consistently applied to biological assessment and BCG development for streams and rivers. These attributes are also being explored for application in larger, more complex systems such as large rivers, estuaries, and coral reefs (see Appendix B). Additionally, these attributes may be more easily assessed, quantifiable, and amenable to rule development using spatial analysis.

Attribute assignment uses both empirical data analysis and expert judgment. Typically, tolerances of many genera or species are available from well-known compendia on macroinvertebrates (e.g., Barbour et al. 1999; Hilsenhoff 1982; Merritt et al. 2008). The published tolerances are broad and might not apply to species or genera in the data set at hand, but they provide a convenient initial value for the panel to consider. To augment the published tolerances and traits information, local data are also evaluated empirically to determine whether the published values, or the expert's opinions, are supported by the local data.

While it is tempting to rely only on the empirical analysis and "let the data tell the story," in practice, many data sets are not sufficient to determine tolerance of all taxa. For example, a taxon that occurs in five samples is too infrequent in the data set to estimate its tolerance. Nevertheless, it may be a taxon where the tolerance is well-established; for example, *Limnodrilus hoffmeisteri* is a worm characteristic of severe organic enrichment associated with untreated sewage discharge, and Brook Trout is a highly sensitive fish species in streams of northeastern North America. Both of these organisms are relatively uncommon in regions of the country with a high degree of development and with regulated discharges.

However, their biology and tolerance are well-known. Similarly, there are likely to be other taxa for which the assembled experts have substantial experience, but that might be insufficiently represented in the data set. Presentation of the stress-response analysis ensures that all experts in the workgroup are aware of, and familiar with, the data set at hand and associations that exist in that data set.

Empirical analysis of the data set being used in the calibration can greatly assist the attribute assignment. After developing a stressor gradient, it becomes possible to support assignment of taxa to attributes based on biological responses to the stressor gradient. This is similar to the analysis often used to identify tolerance groups (e.g., Yuan 2004, 2006).

Several different statistical approaches can be applied to examine individual species' response to stressors: (1) correlation tables and simple scatter plots, (2) central tendencies, (3) environmental limits, (4) optima, and (5) curve shapes (Yuan 2006). Correlations and scatter plots show the strength and shape of a stress-response. Tolerance values expressed in terms of central tendencies attempt to describe the average environmental conditions under which a species is likely to occur; tolerance expressed in terms of environmental limits attempt to capture the maximum or the minimum level of an environmental variable under which a species can persist; and tolerance expressed in terms of optima define the environmental conditions that are most preferred by a given species. These types of tolerances are expressed in terms of locations on a continuous numerical scale that represent the environmental gradient of interest. Both abundance-based and presence/absence-based models can be built using these statistical approaches. See Yuan (2006) for analytical methods.

3.3.2.1 Example: Stressor-response of Macroinvertebrates (Maryland Piedmont) and Fish (Minnesota Lakes)

When panelists assign taxa to attribute groups I–VI, they rely on a combination of empirical examination of taxon occurrences at sites that span a human disturbance, or stress, gradient, as well as professional experience as field biologists who have sampled water bodies in the areas of interest. During the attribute assignment process, panelists are provided with taxon-response plots in which the frequency and abundance of the taxa are plotted over the range of the disturbance gradient (Yuan 2006). Several different statistical models can be used to generate these plots, including:

- Weighted averaging to estimate optima and tolerance values (abundance based).
- Cumulative distribution function median and extreme limits (presence/absence).
- Logistic regression (linear, nonlinear, generalized additive model) median and extreme limits (presence/absence).

Taxon-response plots can be used to infer central tendencies (average environmental conditions under which a species is likely to occur), tolerance limits (maximum or minimum levels of an environmental variable under which a species can persist), and optima (environmental conditions that are most preferred by a given species).

The panelists use these plots to help inform BCG attribute assignments, particularly for attributes II (highly sensitive), III (intermediate sensitive), IV (intermediate tolerant), and V (tolerant). Taxa in these attribute categories are expected to follow the response patterns shown in Figure 11.

Prior to generating the plots, stressor variables are selected based on considerations such as availability of quantitative field-collected data and responsiveness of the biological assemblage to the stressor, or a stressor index such as Minnesota's HDS (Table 6). In one example, taxon-response plots were generated for two stressor variables—imperviousness and habitat index scores—based on data from the Northern Piedmont of Maryland (Stamp et al. 2014). For Minnesota lakes, the group examined taxon-response plots for urban/agricultural/mining land use in the contributing watershed and the trophic state index (Gerritsen and Stamp 2014). Examples of taxon-response plots from these two projects can be found in Figure 12. In these examples, there was good agreement between the taxon-response plots and attribute assignments, but this does not always happen. In cases of disagreement, the group relies on consensus professional opinion, unless contradicted by an overwhelming response in the data analysis. To interpret the graphs in Figure 12, the points are actual data of relative abundance, the curve represents the capture probability (logistic regression generalized additive model fit and confidence interval following Yuan 2006), and the red vertical dashed lines represent the optimum (50%) and tolerance (95%) values. Curves are smoothed to facilitate comparison to the "ideal" plots of Figure 11.

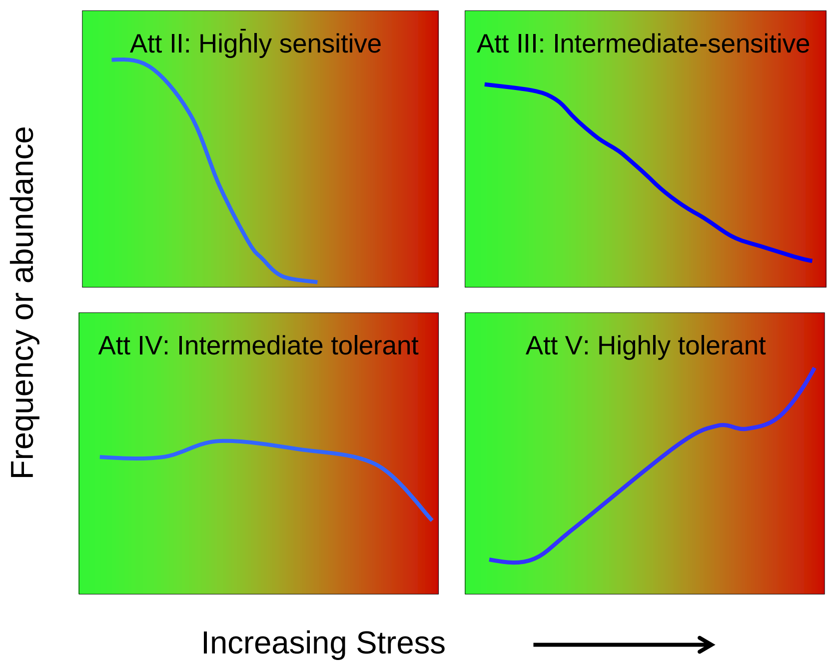

Figure 11. The frequency of occurrence and abundances of attribute II, III, IV, and V taxa are expected to follow these patterns in relation to the stressor gradient. Attribute II taxa have a high relative abundance and high probability of occurrence in minimally-disturbed sites. Attribute III taxa occur throughout the disturbance gradient, but with higher probability in better sites. Attribute IV taxa also occur throughout the disturbance gradient, but with roughly equal probability throughout, or with a peak in the middle of the disturbance range. Attribute V taxa occur throughout the disturbance gradient, but with higher probability of occurrence, and higher abundances, in more stressed sites.

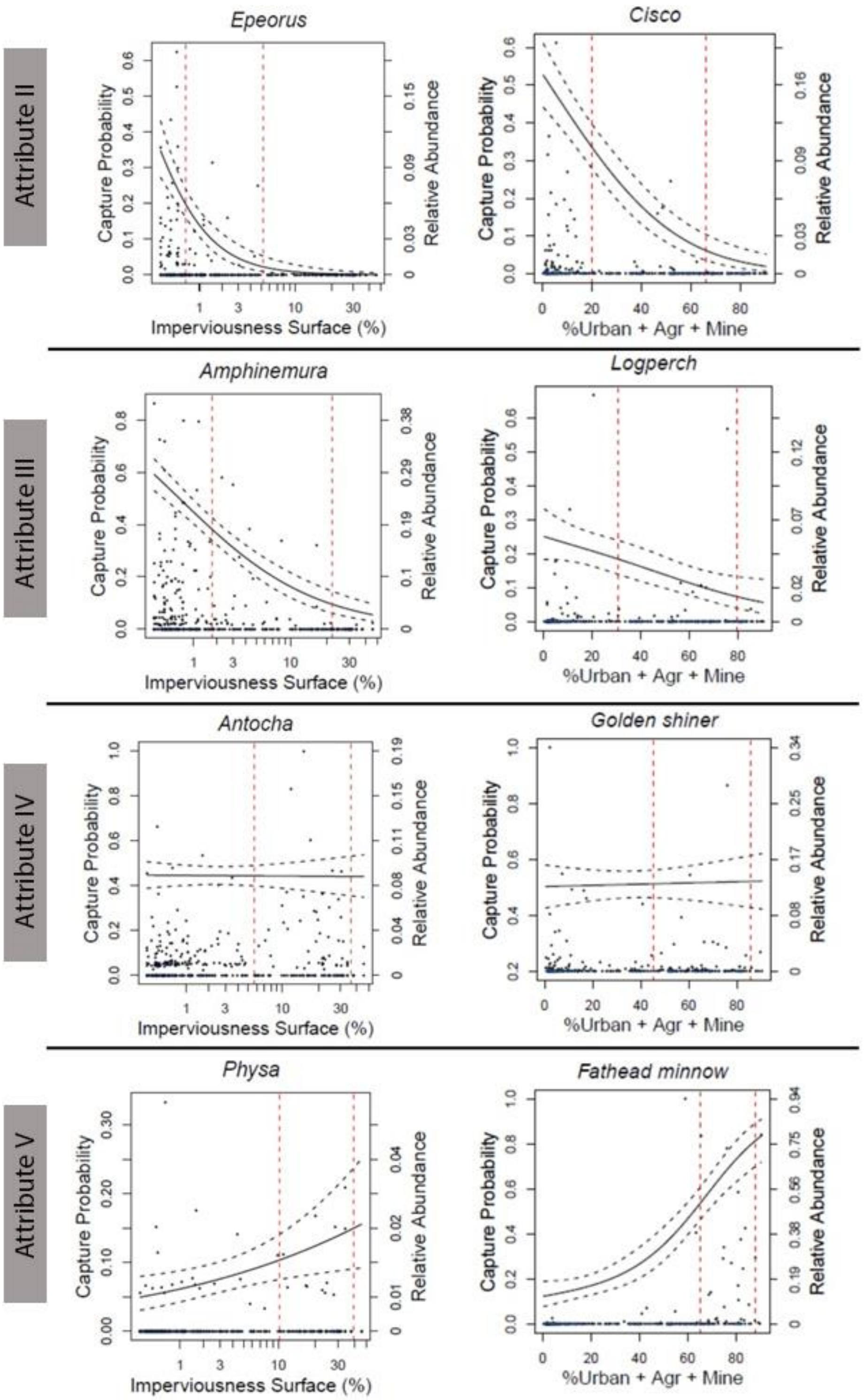

Figure 12. Examples of attribute II (highly sensitive), III (intermediate sensitive), IV (intermediate tolerant), and V (tolerant) taxon-response plots for the Northern Piedmont of Maryland and Minnesota lakes. The plots on the left show responses of four macroinvertebrate taxa from the Northern Piedmont of Maryland to impervious surface (the x-axis is log-transformed). The plots on the right show responses of five fish taxa from Minnesota lakes to urban/agricultural/mining land use.

3.3.3 Data Preparation: Organize Data for Expert Panel

The expert panel will need to work with a taxon list for the database and with sample data. The taxon list should include the taxonomic hierarchy for each genus or species in the database, and it should be sorted taxonomically for ease of use. Information to be included for each species should include known tolerance/sensitivity from other sources (e.g., published Hilsenhoff tolerances, trophic guild, spawning guild, habit, habitat preference). For lists of taxa that include some of these characteristics, see Barbour et al. (1999) and Merritt and Cummins (1996).

The panel will also need to work with data sheets from individual sites. Figure 13 is an example of a data sheet that has been used in stream BCG development. These sheets should include all taxa, counts, and the panel-assigned attribute for each taxon, sorted taxonomically. Attribute assignments (left-hand column, Figure 13) are finalized during the expert panel meeting, and they are entered into the tables at that time.

In a typical workshop, the expert panel should have data available from approximately 20 to 40 sites from a single water body class, which are selected (by data analysts, not panelists) from the entire range of the stressor gradient. There should be good representation of least stressed sites, as well as most stressed sites, and all categories of stress in between. The sites selected are typically a subset of sites used to develop the stress-response curves (Figure 12).

Although the data analysts have selected cover the range of disturbance, stress information on individual sites is not provided to the expert panel. In BCG development, the rating should be done "blind" without knowledge of stressors or levels of disturbance to minimize preconceived perceptions and bias.

BCG_SampID	HA11	Assigned Level	Area (km²)	7.68
StationID			Pct Urban	
Station Name			Pct Agr	
WMA			Pct Forest	
Gradient	High		Pct Wetlands	
CollDate	05-05-1994		Habitat Score	
BCG Attribute	**FinalID**	**Individuals**	**Order**	**Family**
4	*Psephenus herricki*	1	Coleoptera	Psephenidae
2	*Diamesa nivoriunda*	2	Diptera	Chironomidae
5	*Dicrotendipes neomodestus*	1	Diptera	Chironomidae
5	*Orthocladius dorenus*	6	Diptera	Chironomidae
5	*Orthocladius obumbratus*	1	Diptera	Chironomidae
5	*Orthocladius rivulorum*	2	Diptera	Chironomidae
5	*Micropsectra*	1	Diptera	Chironomidae
5	*Tanytarsus*	1	Diptera	Chironomidae
2	*Acentrella turbida*	2	Ephemeroptera	Baetidae
2	*Drunella cornutella*	12	Ephemeroptera	Ephemerellidae
3	*Ephemerella dorothea*	21	Ephemeroptera	Ephemerellidae
3	*Ephemerella rotunda*	3	Ephemeroptera	Ephemerellidae
3	*Eurylophella temporalis*	12	Ephemeroptera	Ephemerellidae
2	*Epeorus*	2	Ephemeroptera	Heptageniidae
2	*Ameletus*	3	Ephemeroptera	Siphlonuridae
3	*Amphinemura delosa*	25	Plecoptera	Nemouridae
2	*Isoperla transmarina*	6	Plecoptera	Perlodidae
4	*Ceratopsyche slossonae*	1	Trichoptera	Hydropsychidae
4	*Cheumatopsyche*	1	Trichoptera	Hydropsychidae
5	*Hydropsyche betteni*	1	Trichoptera	Hydropsychidae
3	*Pycnopsyche*	4	Trichoptera	Limnephilidae
4	*Polycentropus*	1	Trichoptera	Polycentropodidae

Summary

BCG Attribute	Taxa	Individuals
1	0	0
2	6	27
3	5	65
4	4	4
5	7	13
6	0	0
x	0	0
Total	*22*	*109*

Figure 13. Example data table for site assessment, showing how site data may be arranged for a panel's assessment. Attribute summary information is included at the bottom. Note that stressor information is blank—the panel rates sites without knowledge of stressors.

3.4 Step Three: Convene an Expert Panel

The expert workshop to calibrate the BCG is central to BCG development. Calibrating a BCG requires refining the generalized conceptual model to reflect regional conditions (Davies and Jackson 2006). The process has several steps:

- An expert panel of ecologists and field biologists is assembled.
- The panel assigns taxa to attributes I–VI. This step makes use of the taxon-response analysis described in section 3.3, combined with the experience and judgment of panel members.
- The panel assigns a set of sites to levels of the BCG. In this step, the panel also develops a general description of the native aquatic assemblages under natural, undisturbed conditions. The description of natural conditions requires biological knowledge of the region, a natural classification of the assemblages, and, if available, historical descriptions of habitats and assemblages.
- The panel develops narrative and quantitative decision rules to assign sites to BCG levels.

3.4.1 Expert Panel

An expert panel provides specific technical descriptions of each BCG level through the process of assigning sites to the levels. The panel should consist of (1) ecologists with strong field and identification experience with organisms represented in the monitoring data; (2) ecologists with knowledge of the natural history of the organisms and organism tolerances; (3) water quality experts; and, if possible, (4) one or more persons familiar with the historical background and context of water bodies of the region. This expertise could include knowledge of historic vegetation cover of the region and changes to the present or past distributions from museum records and old accounts of the taxa in the species list. Past experience with panels suggests that an ideal number of participants for each assemblage group is between 8 and 12; fewer than 8 results in a narrow diversity of expertise and viewpoints represented, yet a panel with more than 12 participants can become unwieldy and slow in identifying individual opinions. Panel meetings should also include a facilitator familiar with the BCG calibration process; staff familiar with the data and analysis already done (section 3.3); and recorder(s) to record decisions, expert logic, and important discussion points.

In the introductory session of the workshop, the panel is introduced to the BCG concept and ground rules for assessing sites and developing decision rules. Panel members must have sufficient time to digest and discuss the process and feel comfortable with it. This requires one or more introductory sessions to familiarize them with the conceptual BCG model, applications, calibration, and the data and procedures to be used. These sessions may be done as webinars to save time in the face-to-face panel meetings. For several of the BCG development efforts, two to three webinars have been conducted with the expert panel and have proven to be very effective in educating the panelists about the BCG. These webinars have also been useful in addressing questions and issues ahead of the workshop that would otherwise have sidetracked the work of the panel during the face to face meeting. Additionally, a dry run with the panelist in use of data spreadsheets and evaluating the data can result in new information and insight from the panelists that can be incorporated into developing the BCG. A very useful initial exercise is a “practice run” to rate approximately three sites that the facilitation team has reason to believe might be relatively good condition, mediocre condition, and poor condition, respectively. This allows panelists to experience the process on which they will be spending considerable time. Upon completion of the introductory session, the panel begins work, as explained in the following sections.

3.4.2 Assign Taxa to Attributes

Prior to calibrating BCG levels, the panel assigns taxa in the database to the taxonomic attributes (attributes I to VI). Assignments of taxa to attributes rely on examination of empirical stress-response relations, as well as professional experience of field biologists who have sampled the water bodies of the region. In this way, the professional opinions of the workgroup can be tested with the empirical data (Figure 12). Several taxa may have insufficient data within the statewide data set. The wider collective experience of the workgroup can enhance the empirical database with experience with under-represented taxa, and knowledge of natural history.

In cases of disagreement between empirical analyses and professional opinion, the group can employ a weight of evidence approach, including consensus professional opinion and strong and consistent response shown in the data analysis (Figure 12). To save time in the face-to-face panel workshop, attributes and assignment of taxa to the (taxonomic) attributes can be introduced in the pre-workshop webinars, and each expert is asked to assign taxa to attributes as homework. Experts are also given results of the stressor-response analyses of individual taxa. The facilitation team compiles the experts' taxon assignments prior to the workshop, and participants discuss each taxon to develop consensus assignments.

After the taxa are assigned to the attributes, the attribute numbers should be entered into the site-specific data sheets (Figure 12).

3.4.2.1 Example: Alabama Taxon Assignments

Prior to the face-to-face BCG workshop for northern Alabama streams, panelists received taxa lists from the facilitation team and were asked to make preliminary attribute assignments based on (1) relevant literature and (2) taxon-response plots showing relationships between the frequency and abundance of the taxa over the range of the Alabama HDG. The facilitation team compiled the results and used them as a starting point for the attribute assignment component of the workshop, during which panelists made assessments based on consensus professional opinion. Once the attribute assignments were made, the facilitator entered them into a master taxa worksheet, which automatically updated the attribute assignments in the sample worksheets (Figure 13). Table 10 shows the distribution of macroinvertebrate and fish taxa across attribute categories for northern Alabama streams.

Table 10. Distribution of macroinvertebrate and fish taxa across the BCG attributes in northern Alabama

BCG Attribute		Macroinvertebrates # of taxa	Macroinvertebrates % of individuals	Macroinvertebrates Examples	Fish # of taxa	Fish % of individuals	Fish Examples
I	Historically documented, sensitive, long-lived, or regionally endemic taxa	1	0.2	Gastropods: *Fontigens*	6	2.7	Bankhead Darter, Crown Darter, Holiday Darter, Sipsey Darter
II	Highly sensitive taxa	110	16.7	Beetles: *Optioservus*, Mayflies: *Heptagenia*, *Leucrocuta*, Caddisflies: *Brachycentrus, Glossosoma*, Stoneflies: *Leuctra, Tallaperla*	15	6.8	Burrhead Shiner, Cahaba Shiner, Bigeye Shiner, Goldline Darter, Warpaint Shiner, Blenny Darter
III	Intermediate sensitive taxa	136	20.6	Beetles: *Macronychus,* Mayflies: *Stenonema, Isonychia,* Midges: *Tvetnia, Brillia,* Caddisflies: *Chimarra,* Odonata: *Macromia*	38	17.4	Shadow Bass, Black Redhorse, Rock Bass, Northern Studfish, Southern Studfish, Bigeye Chub, Tuskaloosa Darter, Rainbow Shiner
IV	Taxa of intermediate tolerance	173	26.2	Midges: *Polypedilum, Tanytarsus, Rheotanytarsus, Thienemannimyia,* Beetles: *Stenelmis,* Dragonflies: *Boyeria,* Mayflies: *Baetidae*	76	34.7	Longear Sunfish, Alabama Hog Sucker, Banded Sculpin, Alabama Shiner, Silverstripe Shiner
V	Tolerant native taxa	67	10.2	Caddisflies: *Cheumatopsyche,* Worms: *Oligochaeta,* Midges: *Ablabesmyia, Dicrotendipes,* Dragonflies: *Argia,* Flies: *Simulium,* Gastropods: *Physella*	29	13.2	Bluegill, Blackbanded Darter, Largemouth Bass, Striped Shiner, Spotted Bass, Blacktail Shiner, Blackspotted Topminnow
Va	Opportunistic tolerant taxa	—	—	—	9	4.1	Creek Chub, Bluntnose Minnow, Redbreast Sunfish, Western Mosquitofish, Eastern Mosquitofish, Green Sunfish, Largescale Stoneroller, Yellow Bullhead
VI	Non-native taxa	2	0.3	*Corbicula* and Astacidae	5	2.3	Common Carp, Fathead Minnow, Goldfish, Grass Carp, Red Shiner
X	Migrating fish (surrogate for ecosystem connectance)	—	—	—	2	0.9	American Eel, Atlantic Needlefish
—	No attribute assignment (insufficient information)	171	25.9	Coarse identifications and uncommon occurrences	39	17.8	Uncommon occurrences
Totals		***660***	***100***		***219***	***100***	

3.4.3 Assign Sites to Condition Levels

Working from a description of undisturbed communities and the species composition data from example sites, the panel assigns sites to the levels of the BCG. These site assignments are used to describe changes in the aquatic communities for lower levels of biological condition, leading to a complete descriptive model of the BCG for the region. Throughout this process, the panel makes use of the prepared data (Figure 12 and Figure 13) to examine species composition and abundance data from sites with different levels of cumulative stress, from least stressed to severely stressed.

3.4.3.1 Description of Natural, Undisturbed Conditions

First, the panel attempts to reconstruct the native aquatic assemblages under natural, undisturbed conditions. This is an application of historical ecology (McClenachan et al. 2015), and if resources are available, a formal effort should be made to describe the historical conditions. The description of natural conditions requires biological knowledge of the region, a natural classification of the assemblages, and, if available, historical descriptions of the habitats and assemblages. A useful exercise is to ask each panelist to describe the community of an undisturbed, natural system. This develops a best professional judgment description of undisturbed communities for the region that is, at best, qualitative.

Descriptive studies of historic and prehistoric distributions of species can be useful in developing a description of pre-settlement or pre-industrial conditions. For example, most classic fish distribution monographs draw heavily on early descriptions and collections by 19th century naturalists (e.g., descriptions in *The Fishes of Ohio*; Trautman 1981) to develop estimates of pre-settlement distributions for as many species as possible. Fish and mollusks have also been investigated from native and early settler middens to derive distributions of harvested species, and these can be combined with other studies to develop more comprehensive descriptions (e.g., Angelo et al. 2002, 2009).

For example, in Kansas, few streams have completely escaped the effects of large-scale agricultural and livestock practices implemented over the past 150 years (Angelo et al. 2009). Although many of the biological surveys from the mid-1800s were performed after the start of intensive agriculture, they still provide valuable documentation of the occurrence of several freshwater species that soon disappeared from specific watersheds or the region as a whole. Museum collections and other historical records indicate that many creeks and smaller rivers in the Great Plains supported a variety of predominately eastern fish and shellfish species, most requiring clear water and relatively stable stream bottoms. In fact, Kansas was once home to more than 50 Unionid mussel species. Today, several mollusk species are no longer found in most of their original habitats (Figure 14). Over the past 150 years, at least 11 aquatic molluscan taxa have become extinct in Kansas, and an additional 23 species are currently designated as endangered, threatened, or vulnerable.

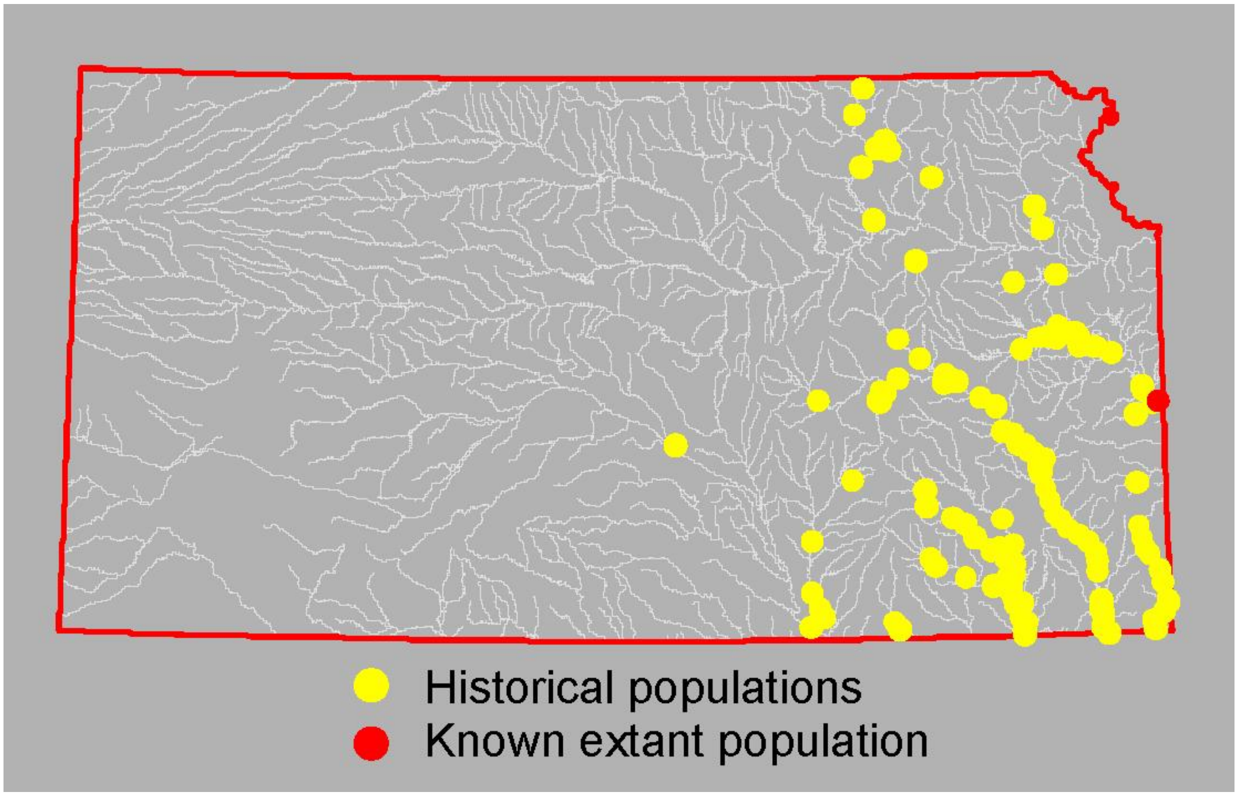

Figure 14. Decline in geographical distribution of black sandshell mussel in Kansas (after Angelo et al. 2009).

A description of undisturbed conditions may also be developed more quantitatively if databases, expertise, and resources are available. With the growth of biological monitoring, there have been several recent attempts to develop predictive statistical models of biological composition (typically metrics, but also taxa) using multiple regression (e.g., Waite et al. 2010) or other modeling approaches (e.g., random forests [DeWalt et al. 2009]; Threshold Indicator Taxa ANalysis (TITAN) [Baker and King 2010]). These model approaches can be used to extrapolate to undisturbed conditions and predict relevant metrics (Waite et al. 2010), composition, or individual species ranges (DeWalt et al. 2009) under undisturbed conditions. They are especially useful if museum records, paleolimnological investigations, or historical descriptions do not apply (e.g., invertebrates were typically of less interest than fish to early explorers and many naturalists).

There are challenges and drawbacks when using historical data to reconstruct natural stream conditions. It takes a great deal of time and commitment to piece together numerous bits of information, especially considering the limitations and inconsistencies inherent in historical data. Much of the information is not directly comparable to modern assessment data, largely because results from previous studies and observations are often based on different sampling methodologies. Sometimes the data are not applicable because they were obtained after settlers significantly impacted the land, but often such physical habitat data are missing or incomplete. Finally, some regions settled early in the history of the nation may simply lack definitive historical data on the baseline biological condition.

As an example, Shumchenia et al. (2015) constructed the first estuarine BCG framework that examines changes in habitat structure through time. Using historical data and descriptions, including maps, navigational charts, land use descriptions, sediment cores, and shellfish landings, they described a minimally disturbed range of conditions for the ecosystem, anchored by observations before 1850. Like many estuaries in the U.S., the relative importance of environmental stressors changed over time, but even qualitative descriptions of the biological indicators' status provided useful information for defining condition levels. In addition to helping conceptually define the biotic community expected in an undisturbed or minimally disturbed environment, the BCG was used to show that stressors rarely acted alone and that declines in one biological indicator influenced the increase or decline of others.

3.4.3.2 Assignment of Current Sites

The panel works with data tables showing the species and attributes for each site (Figure 13). In developing assessments, the panel works "blind," that is, no stressor information is included in the data table. Only non-anthropogenic classification variables are shown (in Figure 13, watershed area and gradient). Sites are selected by the facilitation team to span the range of stress that occurs in the region, from the least stressed to the most stressed. Panel members discuss the species composition and what they expect to see for each level of the BCG.

A typical site assignment proceeds as follows: The facilitator projects the data onto a screen (Figure 13) and calls out some salient data on the site, including area, gradient, total taxa, and possibly some summary metrics. Panelists take several minutes to look at the data, and each panelist proposes a BCG level for the site, along with principal reasons for the decision. The site and decision reasons are discussed by the panel, and panelists are allowed to change their decisions, if desired.

Following assignment of 20 or more sites to levels of the BCG, the panel develops a description of each level, along with rules that are expected to be met by each level, starting from the highest quality condition observed in the data set (e.g., level 1) and working down to the most severely altered condition (e.g., level 6). The description and rules can be as quantitative as the panel cares to make them. Examples of water bodies that might have low resolution include intermittent and ephemeral streams, wetlands, and tidal fresh portions of estuaries. Also, BCG levels might be absent from the data set. In most developed states, there is general recognition that BCG level 1 is exceedingly rare or absent. BCG level 6 is often absent from data sets because the most egregious pollution has been remediated, leaving level 5 as the poorest quality observed. Level 6 may sometimes be observed in older data (pre-1985). If a panel determines that two or more levels cannot be discriminated, then they are typically combined into one; for example "levels 3–4" or "levels 5–6." This should only be done when the panel determines that the levels cannot be discriminated, not simply because one or more levels happen to be absent from the given data set.

Assessing biological condition and assigning sites to a level of the BCG are based on the detailed attribute descriptions developed earlier for the water body and region for which the model is being developed, plus other taxonomic attributes the panel agrees are important. It is entirely possible to determine biological condition with a subset of the attributes. For example, biological assessment in streams and rivers is currently carried out with indicators very similar to taxonomic and condition attributes I through VII of the BCG, all derived from species composition. However, a measure of the spatial distribution of estuarine habitats for assessing whole estuary condition is under development in Narragansett Bay based on a spatial habitat measure and on the "historic balance" of critical estuarine habitats in Tampa Bay (Cicchetti and Greening 2011; Shumchenia et al. 2015). This indicator is under development as a surrogate for attribute X (ecosystem connectivity), and would provide information on the presence and spatial relationship of habitats critical to a functioning estuarine system. The importance of individual attributes depends on the system being assessed, and information or indicators for all attributes may not be necessary.

As an example, a panel of aquatic biologists from three states (Michigan, Wisconsin, and Minnesota) and four tribal water quality agencies calibrated BCG models for coolwater wadeable streams of the Upper Midwest (Gerritsen and Stamp 2012). Prior to performing site assessments, the group discussed their expectations for sites spanning the different BCG levels. Table 11 contains the narrative descriptions of each of the BCG levels (modified after Davies and Jackson (2006)), as well as lists of fish and macroinvertebrate taxa that the group expected to commonly find in samples from each BCG level. The overall relationship between BCG level and Minnesota's disturbance score is shown in Figure 15.

Table 11. Description of transitional cold-cool assemblages (benthic macroinvertebrate and fish taxa) in each assessed BCG level, Upper Midwest coldwater streams. Definitions are modified after Davies and Jackson (2006) (Source: Gerritsen and Stamp (2012)).

BCG level 1	**Definition:** Natural or native condition—*native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability*
	Fish: If the stream is in a location where brook trout are native, *native brook trout* must be present. Non-native salmonids must be absent. Up to twelve additional taxa, including highly sensitive (attribute I, II, & III) species such as *slimy sculpin* and *brook lamprey*, are also be present. If tolerant taxa are present, they occur in very low numbers.
	Macroinvertebrates: There is a lack of sufficient information to know what the historical undisturbed macroinvertebrate assemblage looked like.
BCG level 2	**Definition:** Minimal changes in structure of the biotic community and minimal changes in ecosystem function—*virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability*
	Fish: Overall taxa richness and density is as naturally occurs. Non-native salmonids may be present. If the stream is in a location where brook trout are native, *native brook trout* must be present and must not be negatively impacted by non-native salmonids such as *brown trout*. Other highly sensitive (attribute II) and intermediate sensitive (attribute III) taxa such as *sculpins (mottled or slimy), dace (pearl, finescale, northern red belly, longnose)* and *brook lamprey* are also present. Tolerant taxa may be present but in low numbers.
	Macroinvertebrates: Overall taxa richness and density is as naturally occurs. Most sensitive (attribute II) taxa (e.g., *Trichoptera*: *Glossosoma, Rhyacophila, Lepidostoma, Dolophilodes; Ephemeroptera*: *Ephemerella, Epeorus; Plecoptera: Leuctridae*) and other taxa must be present. These plus intermediate sensitive (attribute III) taxa (e.g., *Ephemeroptera*: *Paraleptophlebia*; *Plecoptera*: *Acroneuria, Isoperla, Paragnetina*; *Trichoptera*: *Brachycentrus, Chimarra*) occur in higher relative abundances than in BCG level 3 samples. Tolerant taxa occur in low numbers.
BCG level 3	**Definition:** Evident changes in structure of the biotic community and minimal changes in ecosystem function—*Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but intermediate sensitive taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system*
	Fish: Overall taxa richness and density is as naturally occurs. Sensitive taxa such as *dace (pearl, finescale, northern red belly, longnose)* and *northern hog suckers* must outnumber tolerant taxa such as *central stonerollers* and *bluegill.* Taxa of intermediate tolerance (attribute IV) such as *white suckers*, *blacknose dace, common shiners, darters (johnny, fantail),* and *creek chub* are common, and some tolerant (attribute V) taxa such as *northern pike, yellow perch,* and *stonerollers* may be present. If extra tolerant taxa such as *green sunfish* and *bluntnose* and *fathead minnows* are present, they occur in very low numbers.
	Macroinvertebrates: Overall taxa richness and density is as naturally occurs. Similar to BCG level 2 assemblage except sensitive taxa (e.g., Ephemeroptera: Paraleptophlebia; Plecoptera: Acroneuria, Isoperla, Paragnetina; Trichoptera: Brachycentrus, Chimarra; Diptera: Diamesa, Eukiefferiella, Tvetenia) occur in lower relative abundance and the most sensitive (attribute II) taxa may be absent. Taxa of intermediate tolerance (attribute IV) (e.g., *Gammarus, Oligochaeta, Simulium; Coleoptera: Optioservus, Stenelmis; Ephemeroptera: Baetis, Stenonema; Trichoptera: Hydropsyche, Cheumatopsyche*) are common, and some tolerant taxa (attribute V) occur in low numbers.

BCG level 4	**Definition:** Moderate changes in structure of the biotic community and minimal changes in ecosystem function—*Moderate changes in structure due to replacement of some intermediate sensitive taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes*
	Fish: Sensitive taxa such as *dace (pearl, finescale, northern red belly, longnose)* and *northern hog suckers* are present but occur in very low numbers. Taxa of intermediate tolerance (attribute IV) such as *white suckers, blacknose dace, common shiners, darters (johnny, fantail)* and *creek chub* are common, and some tolerant (attribute V) taxa such as *northern pike, yellow perch* and *stonerollers* are present. When compared to BCG level 3 samples, highly tolerant taxa such as *green sunfish* and *bluntnose* and *fathead minnows* are present in greater numbers.
	Macroinvertebrates: Overall taxa richness is slightly reduced. Sensitive taxa (including EPT taxa) are present but occur in low numbers. Taxa of intermediate tolerance (attribute IV) (e.g., *Gammarus, Oligochaeta, Simulium; Coleoptera: Optioservus, Stenelmis; Ephemeroptera: Baetis, Stenonema; Trichoptera: Hydropsyche, Cheumatopsyche*) are common, as are tolerant (attribute V) taxa (e.g., *Diptera: Cricotopus, Dicrotendipes, Paratanytarsus; Hyalella; Physa; Turbellaria*).
BCG level 5	**Definition:** Major changes in structure of the biotic community and moderate changes in ecosystem function—*Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials.*
	Fish: Overall taxa richness may be reduced. Sensitive taxa drop out. Taxa of intermediate tolerance (attribute IV) such as *white suckers, blacknose dace, common shiners, darters (johnny, fantail),* and *creek chub* are common. There is an influx of tolerant and highly tolerant taxa such as *bluegill*, *yellow perch, largemouth bass, northern pike, central stonerollers, bluntnose minnows, fathead minnows,* and *green sunfish*.
	Macroinvertebrates: Overall taxa richness is slightly reduced. Sensitive taxa may be absent. Taxa of intermediate tolerance (attribute IV) (e.g., *Gammarus, Oligochaeta, Simulium; Coleoptera: Optioservus, Stenelmis; Ephemeroptera: Baetis, Stenonema; Trichoptera: Hydropsyche, Cheumatopsyche*) and tolerant (attribute V) taxa (e.g., *Diptera: Cricotopus, Dicrotendipes, Paratanytarsus; Hyalella; Physa; Turbellaria*) are common. Tolerant taxa occur in higher abundances than in BCG level 4 samples.

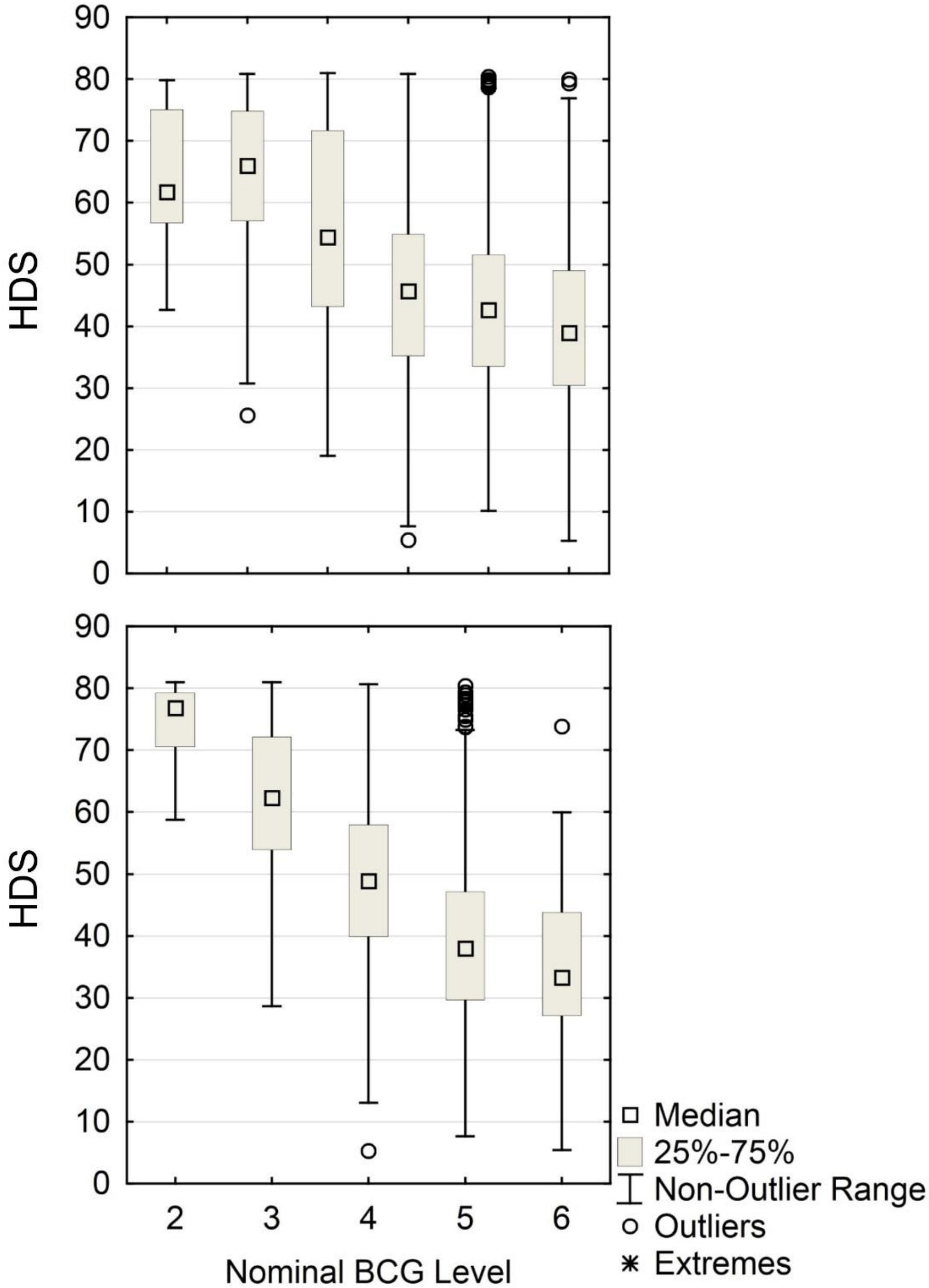

Figure 15. Box plots of HDS for Minnesota streams, grouped by nominal BCG level (panel majority choice) for fish (upper) and macroinvertebrate (lower) samples. HDS scores range from 0 (most disturbed) to 81 (least disturbed) (Gerritsen et al. 2013).

3.4.3.3 Variability in Panelist Biological Condition Gradient Calls

Consistency among panelists is important. In addition to integer BCG levels (e.g., levels 2, 3, 4), panelists also aim to identify sites somewhat better or somewhat worse than the integer levels, up to and including samples that are borderline between adjacent BCG levels. In calibration exercises, intermediate levels have been assigned (+) and (-). This information has been used to help define the threshold where an expert would assign a site to a different BCG level. An expert assigning a site to a BCG level with a (+) or (-) caveat would be asked what additional change in the site data would lead to a different level assignment, and why.

For the BCG project in the Northern Piedmont of Maryland, the macroinvertebrate workgroup assessed 46 calibration samples. Panelists rated samples in the six BCG levels, and modified those with (+) and (-) as desired. Median BCG level assignments were calculated for each sample as the group nominal level.

Deviations of each panelist's assignments from the group median call were estimated, where deviations were assumed to be in quantities of ⅓ BCG level. Deviations are shown in Figure 16. On average, 62% of BCG level assignments matched exactly with the median, 32% were within ±⅓ BCG level, 5% were within ±⅔ BCG level, and 1% differed by one BCG level (Figure 16).

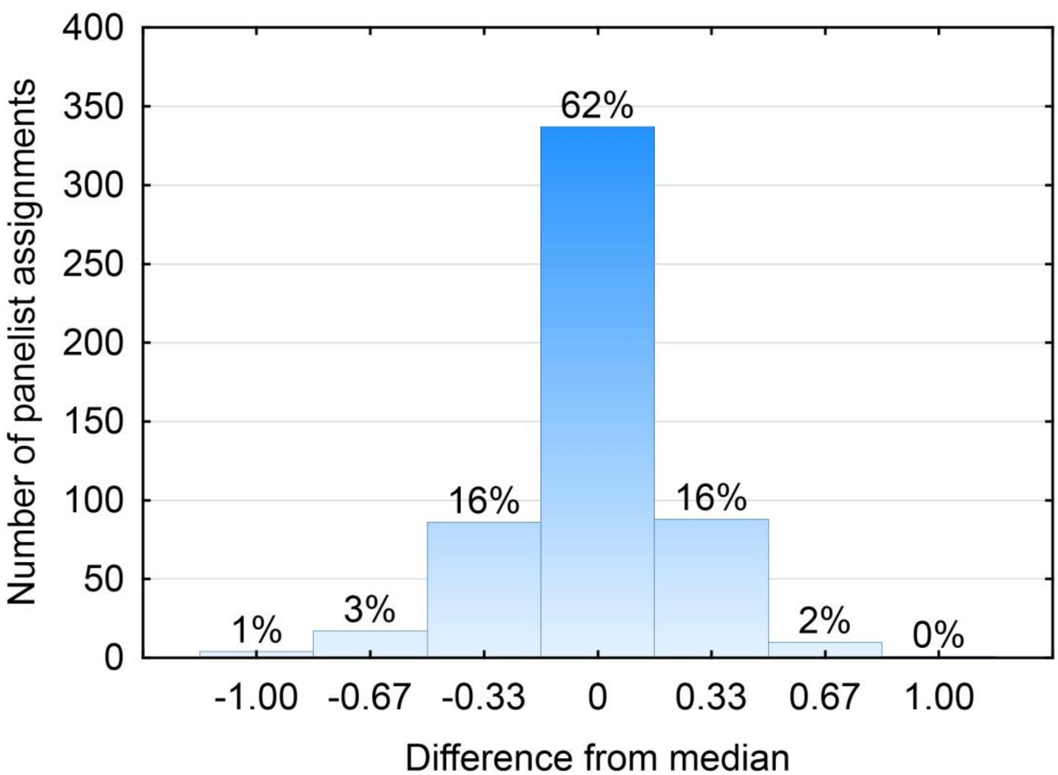

Figure 16. Distribution of individual panelists BCG assignments, as deviations from group sample median, Maryland Piedmont BCG workshop. Percentages above each bar. Data from Stamp et al. 2014.

3.5 Biological Condition Gradient Decision Rules

This chapter described steps to develop narrative descriptions and rules for assigning sites to BCG levels. The core objective of the panel process is to elicit expert judgment on what the experts consider ecologically significant change in the biotic community—and to document the underlying rationale. Through development of expert consensus, first narrative and then quantitative rules emerge, and they are tested and refined based on the current state of the science. Additionally, where gaps in information are identified, the development of decision rules is comparable to formulating a hypothesis, thereby setting up opportunities for applied research that clearly articulate water quality management information needs for goal setting and condition assessments.

The chapter concludes with development of narrative descriptions of BCG levels for specific water bodies within a region or basin. Chapter 4 addresses how to convert these narrative descriptions into narrative then quantitative decision rules for a numeric BCG model. There is no bright line between development of the narrative description and numeric decision rules. In all BCG development efforts to date, preliminary quantitative decision rules have emerged early as part of developing the narrative description and rules. In the first round of data analysis and interpretation, the experts typically formulate their reasoning in the following manner: "I expect more (or fewer) species because …." or "the presence of two or more taxa of attribute III signifies this condition level to me because …." By the second or third round of the data exercise assigning sites to BCG levels, increasingly quantitative statements are provided when experts are asked to explain their logic for assigning sites to BCG levels. These preliminary quantitative statements provide a template for building quantitative decision rules through an iterative, interactive process with the expert panel. Encapsulation of expert judgment provides the transparency and clarity for decision makers and stakeholders to understand the logic and science underpinning ALU goal descriptions and assessments.

Chapter 4. Quantitative Rules and Decision Systems

Routine use of a quantitative BCG model requires a way to automate application of the decision rules so that assessments can be made for newly sampled water bodies without reconvening the expert panel. This chapter discusses approaches to quantify the narrative BCG model and to test and validate the numeric model, corresponding to Steps 4 and 5 of the BCG Calibration process (Figure 8). Quantitative rules rely on sample data using standardized protocols (i.e., most applicable to attributes II–VI). This chapter presents:

- An approach to quantify the conceptual BCG framework and develop a numeric model. This approach is based on elicitation of the experts' decision criteria and incorporation into a numeric decision model using a mathematical set theory approach (e.g., fuzzy logic) (See section 4.1). This approach has been tested and refined in most of the BCG projects to date.
- Considerations and approaches for relating the BCG with the state's existing biological assessment methods and tools (e.g., biological indices such as MMIs and O/E models) (See section 4.2). To date, most states have developed biological indices.
- An additional approach to quantify the BCG narrative decision rules that has been implemented by a state, multivariate linear discriminant modeling. This approach involves development of statistical models that "predict" (or imitate) the expert decisions and may or may not use elicited expert reasoning or rules (See section 4.3). As BCG development and calibration continues, it is expected that the BCG process will be refined and expanded and alternate methods identified and tested.

4.1 Quantitative Rule Development and Application

This approach assumes that because the expert panelists largely agree on BCG ratings for water bodies, they use a common set of decision criteria to achieve the ratings. The approach consists of deriving narrative and numeric decision rules based on expert logic and consensus, including testing of the rules with the expert panel and then with experts outside of the panel. Application of the decision criteria—a set of quantitative rules—can then be applied to any relevant data set or sample.

Quantitative rule and direct decision model development is comprised of the following steps:

- **Elicitation of numeric decision criteria**—During the expert panel meeting, experts are asked for their reasoning behind the decisions. The reasoning is the basis for the BCG level descriptions (Table 11), and also for decision criteria (narrative rules) that the experts use. The narrative rules are elicited from the panel and then quantified.
- **Quantification and testing**—Quantitative rules in turn form the basis of a decision model. A methodology to apply the elicited rules is through a mathematical set theory approach, fuzzy logic (Zadeh 1965, 2008), which mimics human thinking and decision making. Results of the quantitative decision model are compared to the panel's decision, and mismatches are further discussed by the panel to resolve ambiguous or incomplete rules. Ideally, the final model should be tested with an independent data set that was assessed by the panel but not used to calibrate the model. Other approaches to rule elicitation and development include reproducing the expert panel results (but not necessarily their reasoning) with an empirical discriminant analysis model (section 4.2; Davies et al. In press; Shelton and Blocksom 2004), or developing a Bayesian predictive model from the elicitation of reasoning (e.g., Kashuba et al. 2012).

4.1.1 Elicitation of Numeric Decision Criteria

Level descriptions in the BCG conceptual model are intentionally general (e.g., reduced richness, increased dominance, loss or replacement of specific assemblages), which allows for different methods, sources of information, and interpretations to be used in rule development. To allow for consistent assignments of sites to levels, it is necessary to formalize the expert knowledge by codifying level descriptions into a set of rules (e.g., Droesen 1996). If formalized properly, water quality management program scientists with adequate data can follow the rules to obtain the same level assignments as the group of experts. This replicability makes the actual decision criteria transparent to stakeholders.

Rules are logic statements that experts use to make their decisions (e.g., "If plecoptera richness is high, then biological condition is high."). Rules on attributes can also be combined (e.g., "If the proportion of highly sensitive taxa (attribute II) is high, the proportion of tolerant individuals (attribute V) is low, and so on, then assignment is BCG level 2.").

Numeric rule development requires discussion and documentation of level assignment decisions and the reasoning behind the decisions. During this discussion, it is necessary to record each participant's level decision (i.e., vote) for the site, the critical or most important information for the decision (e.g., the number of taxa of a certain attribute, the abundance of an attribute, the presence of indicator taxa), and any confounding or conflicting information and how this information was reconciled for the eventual decision.

As the panel assigns example sites to BCG levels, the panel members are polled on the critical information and criteria they used to make their decisions. These form preliminary, narrative rules that explain how panel members make decisions. For example, "For BCG level 2, sensitive taxa must make up at least half of all taxa in a sample." The decision rule for a single level of the BCG does not always rest on a single attribute (e.g., highly sensitive taxa) but may include other attributes as well (intermediate sensitive taxa, tolerant taxa, indicator species, organism condition), so these are termed "Multiple Attribute Decision Rules." With data from the sites, the rules can be checked and quantified. For mathematical fuzzy set modeling, quantification of rules will allow the agency to consistently assess sites according to the same rules used by the expert panel, and it will allow a computer algorithm, or other persons, to obtain the same level assignments as the panel.

Rule development requires discussion and documentation of BCG level assignment decisions and the reasoning behind the decisions. During this discussion, the facilitators record:

- Each participant's decision for the site:
  - The critical or most important information for the decision—for example, the number or abundance of taxa of a certain attribute, the presence of indicator taxa, the absence of certain taxa, and explanation why this information is ecologically important.
  - Any confounding or conflicting information and how this was resolved for the eventual decision.
- Iteration
  - Rule development is iterative, and it usually requires at least two panel sessions.
  - Building from the initial site assignments, preliminary narrative rules are developed. Descriptive statistics of the attributes and other biological indicators for each BCG level

determined by the panel are then developed for testing. These statistical descriptions will be used for testing and refinement as numeric decision rules are developed and vetted.

- Following the initial development phase, the draft rules are tested by the panel with new data to ensure that new sites are assessed in the same way. The new test sites should not have been used in the initial rule development and also should span the range of anthropogenic stress. Any remaining ambiguities and inconsistencies from the first iterations are also resolved at this stage.

4.1.2 Codification of Decision Criteria: Multiple Attribute Decision Criteria Approach

The expert rules can be automated in Multiple Attribute Decision Models. These models replicate the decision criteria of the expert panel by assembling the decision rules using logic and set theory, in the same way the experts used the rules. In the case studies presented later in this chapter, the models replicated expert panel's decisions at greater than 90% accuracy, including tied or intermediate decisions between adjacent BCG levels (e.g., between level 3 and level 4).

Instead of a statistical prediction of expert judgment, this approach directly and transparently converts the expert consensus to automated site assessment. The method uses modern mathematical set theory and logic (called "fuzzy set theory") applied to rules developed by the group of experts. Mathematical fuzzy set theory is directly applicable to environmental assessment, it has been used extensively in engineering applications worldwide (e.g., Demicco and Klir 2004), and environmental applications have been explored in Europe and Asia (e.g., Castella and Speight 1996; Ibelings et al. 2003).

Mathematical fuzzy set theory allows degrees of membership in sets, and degrees of truth in logic, compared to all-or-nothing in classical set theory and logic. Membership of an object in a set is defined by its membership function, a function that varies between 0 and 1. One can compare how classical set theory and fuzzy set theory treat the common classification of sediment, where sand is defined as particles less than or equal to 2.0 mm diameter, and gravel is greater than 2.0 mm (Demicco and Klir 2004). In classical "crisp" set theory, a particle with diameter of 2.00 mm is classified as "sand," and one with 2.01 mm diameter is classified as "gravel." In fuzzy set theory, both particles have nearly equal membership in both classes (Demicco 2004). Measurement error of 0.005 mm in particle diameter greatly increases the uncertainty of classification in classical set theory, but in fuzzy set theory a particle near the boundary would have nearly equal membership in both sets "sand" and "gravel." Fuzzy sets, thus, retain the understanding and knowledge of measurements close to a set boundary, which is lost in classical sets.

Demicco and Klir (2004) proposed four reasons why mathematical fuzzy sets and logic enhance scientific methodology, and these are applicable to BCG development:

- Fuzzy set theory has greater capability to deal with "irreducible measurement uncertainty," as in the sand/gravel example above.
- Fuzzy set theory captures vagueness of linguistic terms, such as "many," "large," or "few."
- Fuzzy set theory and logic can be used to manage complexity and computational costs of control and decision systems.
- Fuzzy set theory enhances the ability to model human reasoning and decision making, which is critically important for defining thresholds and decision levels for environmental management.

4.1.2.1 Rule-based Inference Model

People tend to use strength of evidence in defining decision criteria, and in allowing some deviation from their ideal for any individual attributes, as long as most attributes are in or near the desired range. For example, the definitions of "high," "moderate," "low," etc. are quantitative and can be interpreted and measured to mean different things. An important step in the BCG process is development of expert consensus defining these, or other, general terms and documenting the expert logic that is the basis for the decisions. The decision rules preserve the collective professional judgment of the expert group and set the stage for the development of models that can reliably assign sites to levels without having to reconvene the same group. In essence, the rules and the models capture the panel's collective decision criteria.

An inference model is developed to replicate the panel decision process, and this section describes an inference model that uses mathematical fuzzy logic to mimic human reasoning. Each linguistic variable (e.g., "high taxon richness") must be defined quantitatively as a fuzzy set (e.g., Klir 2004). A fuzzy set has a membership function, and example membership functions of different classes of taxon richness are shown in Figure 17. In this example (Figure 17), piecewise linear functions (functions consisting of line segments) are used to assign membership of a sample to the fuzzy sets. Fuzzy membership functions were assumed to be adequately defined by piecewise linear functions. Metric values below a lower threshold have membership of 0; values above an upper threshold have membership of 1, and membership is a straight line between the lower and upper thresholds. For example, in Figure 17 (top), a sample with 20 taxa would have a membership of approximately 0.5 in the set "Low to Moderate Taxa" and a membership of 0.5 in the set "Moderate Taxa."

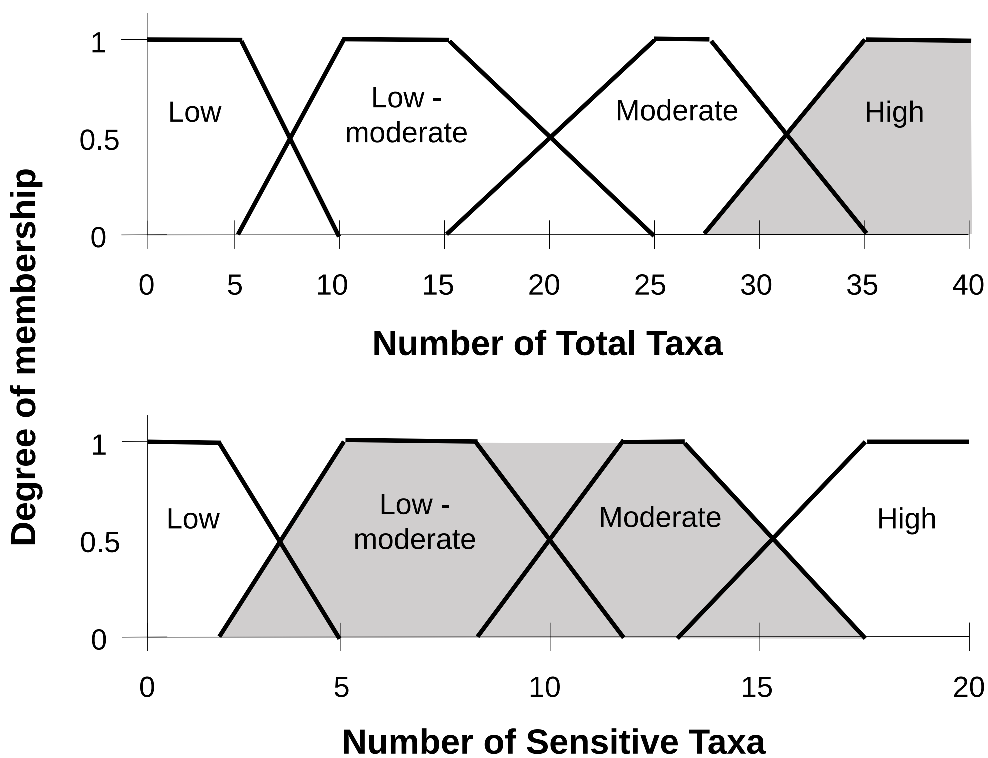

Figure 17. Fuzzy set membership functions assigning linguistic values to defined ranges for Total Taxa (top) and Sensitive Taxa (bottom). Shaded regions correspond to example rules for BCG level 3: "Number of total taxa is high," and "number of sensitive taxa is low-moderate to moderate."

How are inferences made? Suppose there are two rules for determining whether a water body is BCG level 3 (using definitions of Figure 17):

- The number of total taxa is high.
- The number of sensitive taxa is low-moderate to moderate.

In classical set theory, the boundaries between the categories would be vertical lines at the intersections of the membership functions in Figure 17. The rules would then be:

- Total taxa > 30
- Sensitive taxa > 4 and sensitive taxa < 15

If the two rules are combined with an "AND" operator, that is, both must be true, then under classical set theory, if total taxa = 30 and sensitive taxa = 5, the sample would be judged not to be in the set of BCG level 3, because the rule specifies total taxa must be greater than 30. Finding a single additional taxon would result in assessment of BCG level 3. In fuzzy set theory, an AND statement is equivalent to the minimum membership given by each rule:

Level 3 = MIN (total taxa is high, sensitive taxa is low to moderate)

For 30 total taxa, fuzzy membership in "total taxa is high" = 0.5 (Figure 17), and fuzzy membership in "Sensitive taxa is low-moderate to moderate" = 1.0 (Figure 17). Membership of level 3 is then 0.5. In the fuzzy set case, a single additional taxon raises the membership in BCG level 3 from 0.5 to 0.6.

If the two rules are combined with an "OR" operator, then either can be true for a site to meet BCG level 3, and both conditions are not necessary. Crisp set theory now yields a value of "true" if total taxa = 32 and sensitive taxa = 4 (total taxa > 27, therefore it is true). Fuzzy set theory yields a membership of 1 (maximum of 0.5 and 1). Using the fuzzy set theory model, finding a single additional taxon in a sample does not cause the assessment to flip to another level, unlike crisp decision criteria.

Output of the inference model may include membership of a sample in a single level only, ties between levels, and varying memberships among two or more levels. The level with the highest membership value is taken as the nominal level.

4.1.2.2 Quantitative Model Development

Rules identified by the panel, whether quantitative or qualitative, are compared to data summaries of the panel decisions. In particular, if the panel identified a moderate number of sensitive taxa for BCG level 3, then the analyst (i.e., the individual who develops the quantitative decision model) examines the number of sensitive taxa in samples the panel assigned to BCG level 3. The analyst selects a reasonable minimum of the distribution of sensitive taxa in BCG level 3, say the minimum or a 10th quantile, as the decision threshold. This is repeated for all rules and attributes identified by the panel members as being important to their decisions. As a starting point, a plot of the attribute or metric values as box plots by the panel-designated BCG level can be helpful (see section 4.1.2.3 for an example). This type of graphic shows minimum, maximum, median, and selected quantiles for each metric and BCG level. Sample sizes for each BCG level might be small, especially for the highest and lowest levels (BCG levels 1 and 2, and 6, respectively), and might require more professional judgment from the panel to develop rules.

For a particular attribute or metric, the threshold identified by the panel will typically be the 50% membership value in a fuzzy membership function. For example, if the panel identifies "5 or more" sensitive taxa as a requirement for BCG level 3, then 5 taxa would correspond to 50% membership; 3 taxa may correspond to 0% membership, and 7 taxa to 100%. Because number of taxa are always whole numbers, the membership function is not continuous. Some rules are non-fuzzy: if a rule requires "at least 1" or "presence," then presence receives a membership of 100% and absence receives 0%.

A spreadsheet is convenient for developing the rule-based model. Membership functions and rules for each level and each relevant attribute or metric are laid out in the top row, and data for each sample are arrayed in rows. Sample data are called by the rule formulas and the final decision logic is applied to determine membership in each BCG level for each sample.

In models developed up to now, rules work as a logical cascade from BCG level 1 to level 6. A sample is first tested against the level 1 rules; if a required rule fails, then the level fails, and the assessment moves down to level 2, and so on (Figure 18). Depending on how the expert panel makes decisions and rates samples, component rules for a single level may be (1) all-or-nothing (i.e., all rules must be met); (2) some rules have alternate rules (e.g., a very low percentage of tolerant individuals may substitute for a high percentage of sensitive individuals); or (3) any number n of, say, n + 1 rules must be met. Required rules must be true for a site to be assigned to a level. BCG levels 1 and 2 represent minimally-disturbed, natural conditions, hence the rules tend to be the most restrictive. As assemblages change with increasing anthropogenic influence, the changes may manifest in different effects (decline of sensitive species; and/or increases in abundance or dominance of tolerant individuals), and the rules for the middle levels may have more alternative situations. In the more degraded levels (especially BCG level 5), the rules tend to be simple, reflecting a degraded and simplified assemblage. In the cascading logic from BCG level 1 to 6 (Figure 18), there are no rules for level 6 because it is the bottom "bin" that catches sites that fail rules from levels 1 to 5. Examples of these are shown in the case studies that follow.

Two examples on development of numeric decision rules for streams and wadeable rivers follow. The first example shows development of numeric decision rules for benthic macroinvertebrates and fish for cold- and cool-water streams in the Upper Midwest. The second example highlights use of diatom assemblage data from Northern New Jersey in developing a numeric BCG. Both examples illustrate the BCG development process. Macroinvertebrates follow the classic paradigm that overall species richness is higher in the higher BCG levels (levels 1 and 2), but coldwater fish and diatoms are nearly opposite: overall richness is low in pristine coldwater streams, and diatom richness is low in undisturbed oligotrophic streams. Both are dominated by a small number of highly sensitive taxa. As streams become more disturbed, richness and abundance of intermediate and tolerant taxa increase for both fish and diatoms. In the fish assemblage, sensitive taxa disappear in the most disturbed sites, but sensitive taxa may hang on in highly-disturbed diatom assemblages.

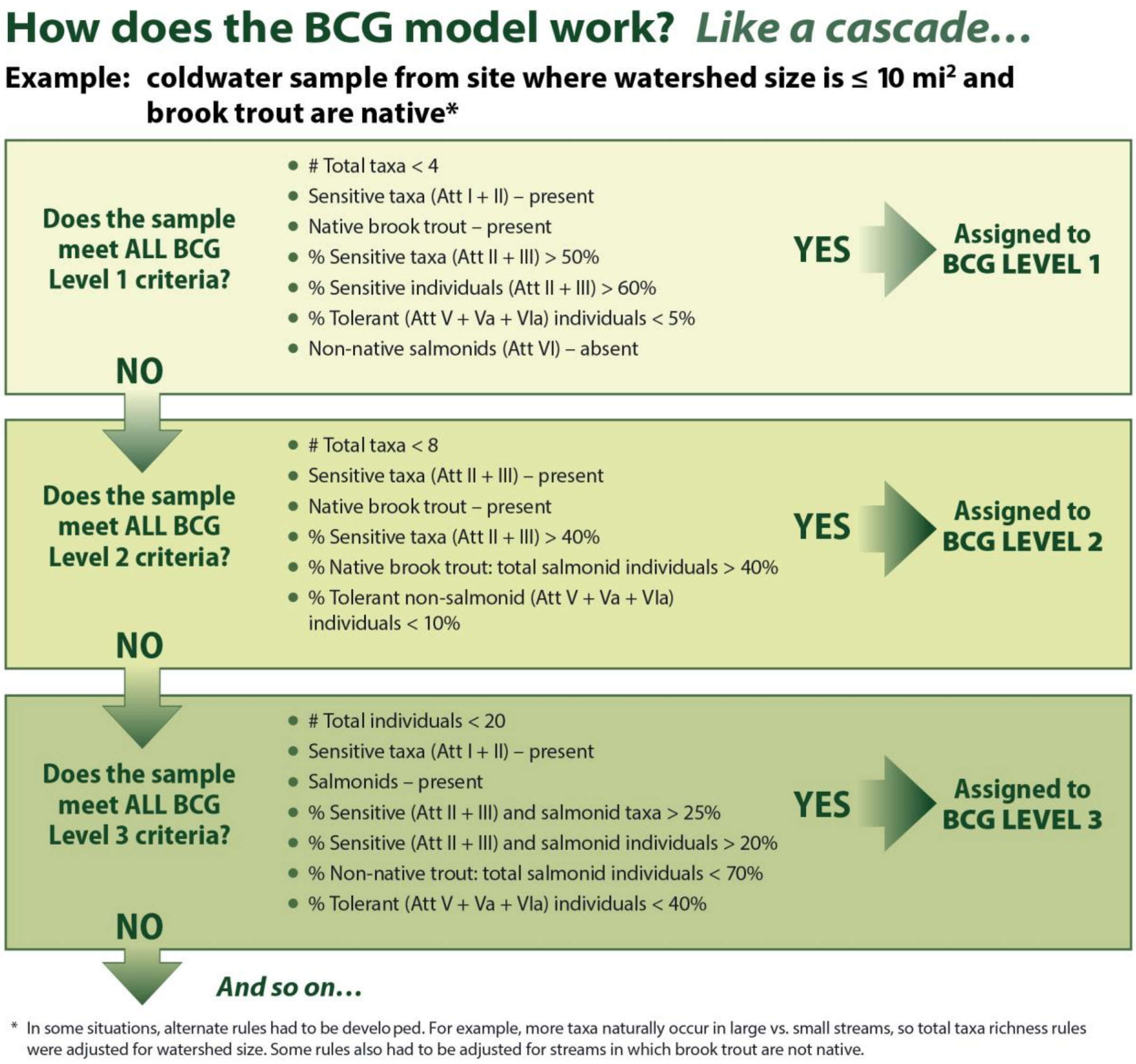

Figure 18. Flow chart depicting how rules work as a logical cascade in the BCG model, from Upper Midwest cold and coolwater example (Source: Modified from Gerritsen and Stamp 2012). For convenience, midpoints of membership functions (50% value) only are shown. For complete rules, see Table 15 and Table 16.

4.1.2.3 Example #1: Quantitative Rules and Decision System for Benthic Macroinvertebrates and Fish, Upper Midwest

Panelists from Indian Nations and the states of Michigan, Wisconsin, and Minnesota calibrated BCG models for fish and macroinvertebrate assemblages in cold and cold-cool transitional wadeable streams of the Upper Midwest (Gerritsen and Stamp 2012). The cool-transitional water macroinvertebrate BCG model was calibrated based on assessments of 37 samples. Panelists made the site assessments using worksheets that contained lists of taxa, taxa abundances, BCG attribute levels assigned to the taxa, BCG attribute metrics, and limited site information, such as watershed area, stream size, average July temperature, and percent forest.

Study Sites

Panelists assigned fish and macroinvertebrate samples from cool-transitional streams to four BCG levels (BCG levels 2–5). Samples were not assigned to BCG level 1 because panelists did not feel that there was enough information to know what the historical undisturbed macroinvertebrate assemblage in this region looked like. Only two of the 37 calibration samples were assigned to BCG level 5 (many of the coolwater sites in this region are in the Northern Lakes and Forests ecoregion). A detailed verbal description of each level is given above in Table 11 (Chapter 3).

Decision rules were initially derived from discussions with the panelists on why individual sites were assessed at a certain level. Panelists made statements such as "BCG level 2 samples should have both a moderate abundance and richness of sensitive taxa (attributes I, II, and III)." These statements were compiled into a set of narrative rules (Table 12).

Table 12. Example of Narrative rules for transitional cold-cool assemblages in Upper Midwest streams (Source: Gerritsen and Stamp (2012))

BCG level 2	**Definition:** Minimal changes in structure of the biotic community and minimal changes in ecosystem function—*virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability*
	Fish Taxa richness is low to moderate Brook Trout, if native, are present Total sensitive taxa are one third of taxa richness Abundance of sensitive individuals is low to moderate Brook Trout (if native) are nearly half of all Salmonidae individuals Tolerant individuals may be a small fraction of total
	Macroinvertebrates Taxa richness is moderate to high Highly sensitive (attribute I and II) taxa make up a very small fraction (or more) of total richness and total abundance All sensitive taxa (attributes I + II + III) make up moderate fraction of richness and abundance Sensitive EPT taxa make up at least a small fraction of total richness
BCG level 3	**Definition:** Evident changes in structure of the biotic community and minimal changes in ecosystem function—*Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa, but intermediate sensitive taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system*
	Fish Taxa richness is moderate but not high Total number of sensitive taxa is greater than tolerant taxa, OR number of sensitive individuals is twice greater than number of tolerant individuals Single most dominant intermediate taxon (attribute III) is less than half of all individuals Extremely tolerant individuals are a very small fraction of total
	Macroinvertebrates Taxa richness is moderate to high Highly sensitive (attribute I and II) taxa are present Total sensitive taxa (attributes I + II + III) make up small fraction of richness and abundance Most dominant tolerant taxon is less than a small fraction of abundance Sensitive EPT taxa make up at least a small fraction of total richness

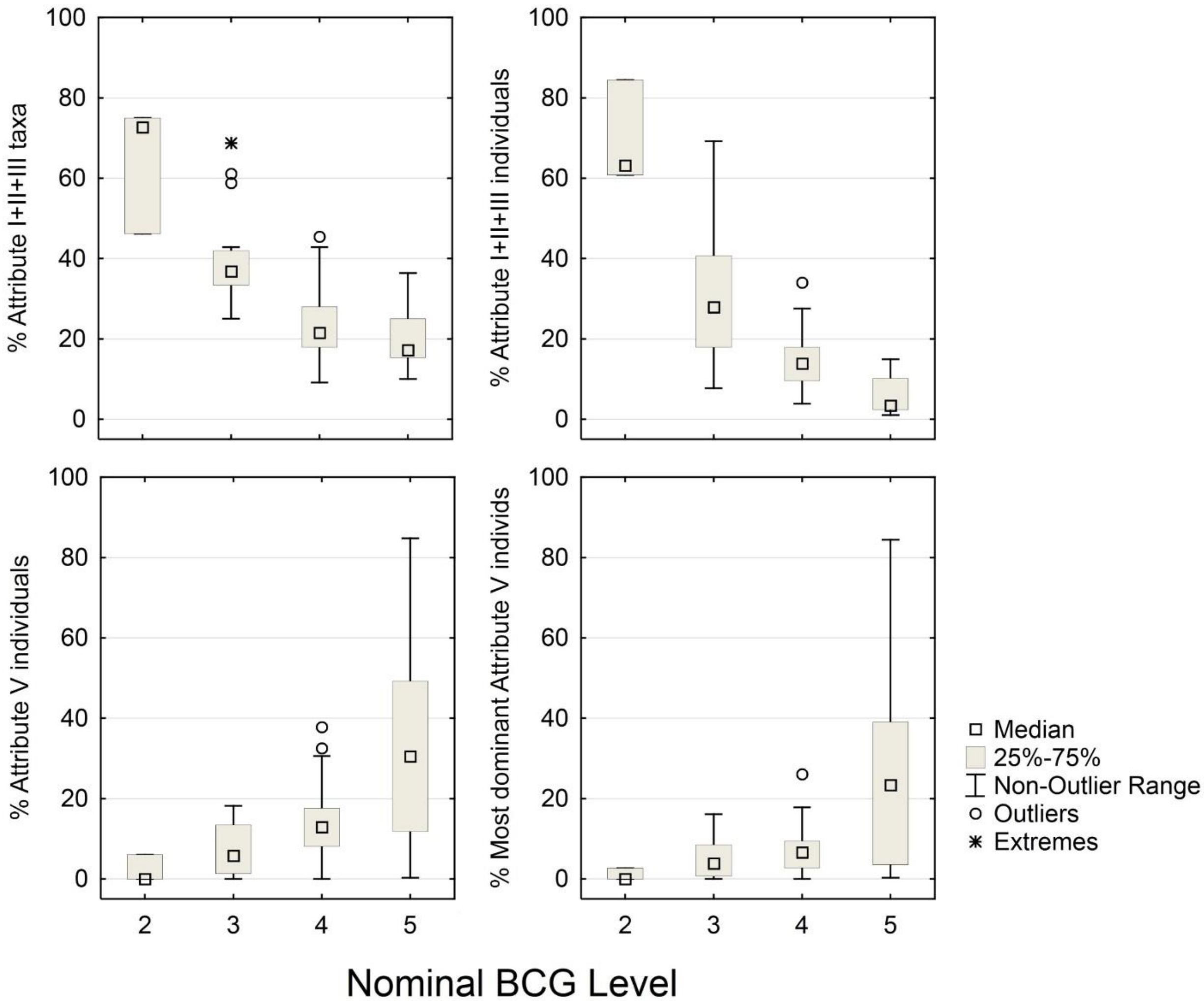

Figure 19. Benthic Macroinvertebrate Taxa: Box plots of sensitive (attribute I+II+III) and tolerant (attribute V) BCG attribute metrics, grouped by nominal BCG level (panel majority choice). These metrics were used in the macroinvertebrate BCG model for coldwater streams in the Upper Midwest.

Using the narrative rules, data were examined for numerical ranges and relationships. For example, examination of the data ranges of attribute I, II, and III taxa for macroinvertebrates (Table 13; Figure 19) showed that the median percent abundance of attribute I, II, and III taxa from BCG level 2 was 75%. The decision rules were adjusted by the empirical distributions of the attribute metrics shown in Table 13 and Figure 19, so that the model would replicate the panel's actual decisions as closely as possible. For the macroinvertebrates, the most important considerations expressed by the experts were percent individuals and percent taxa metrics for attribute II, II+III, IV, and V taxa, and metrics pertaining to three sensitive orders of aquatic insect taxa (e.g., Ephemeroptera, Plecoptera, and Trichoptera (EPT)). Panelists expected BCG level 2 samples to have a moderate presence of highly sensitive (attribute II) taxa, moderate to high total taxon richness, and a low proportion of tolerant (attribute V) taxa. BCG level 3 samples had similar numbers of total taxa but slightly reduced numbers of highly sensitive (attribute II) taxa. Total sensitive taxa (attribute II+III) were still required to be present in BCG level 4 samples, but with reduced richness and abundance. Higher proportions of tolerant (attribute V) individuals occurred in BCG level 4 samples, but could not comprise more than 60% of the assemblage.

BCG level 5 samples were discriminated from BCG level 4 samples by complete loss of sensitive taxa and a further increase in the percent tolerant (attribute V) individuals.

Table 13. Benthic macroinvertebrate taxa: Ranges of attribute metrics in cold-cool transitional macroinvertebrate samples. BCG levels by panel consensus, in the Upper Midwest BCG data set (Gerritsen and Stamp 2012).

Attributes	Metric	BCG Level (Panel Consensus)				
		2 (n=19)	3 (n=13)	4 (n=7)	5 (n=2)	6 (n=1)
0 General	Total Taxa	20–63	20–64	13–58	31–56	31
	Total Individuals	91–359	134–407	138–336	294–321	192
II Highly sensitive taxa	# Taxa	3–11	0–7	0–1	0	4
	% Taxa	8–28	0–15	0–3	0	13
	% Individuals	6–42	0–7	0–1	0	34
III Intermediate sensitive taxa	# Taxa	6–19	7–19	4–17	2–6	16
	% Taxa	19–61	18–49	9–31	6–11	52
	% Individuals	13–55	17–54	3–83	1–9	44
II + III All sensitive taxa	# Taxa	10–26	10–24	4–17	2–6	20
	% Taxa	30–71	22–57	11–31	6–11	65
	% Individuals	31–76	20–56	3–83	1–9	78
	SensEPT # Taxa	6–20	6–14	1–6	2–4	13
	SensEPT_% Individuals	18–71	14–47	2–17	1–2	60
IV Intermediate tolerant taxa	# Taxa	7–28	7–29	8–32	16–29	9
	% Taxa	26–49	35–53	50–65	52	29
	% Individuals	23–53	43–71	17–87	26–30	21
	% Most Dom Individuals	6–31	8–34	5–27	5–15	7
V Tolerant taxa	# Taxa	0–10	1–11	0–9	9–11	0
	% Taxa	0–17	3–22	0–16	20–29	0
	% Individuals	0–22	0–12	0–59	40–72	0
	% Most Dom Individuals	0–17	0–6	0–57	17–59	0

Observations of the attribute metrics from the fish assemblage are shown in Table 14. No attribute I species were identified in the coldwater fish assemblage. The fish assemblage in undisturbed or minimally disturbed coldwater streams typically has few species: native trout, sculpins, and possibly a minnow species. Increases in fish taxa richness in true coldwater is an indicator of degradation. BCG levels 1 and 2 required native trout (Brook Trout), but the native trout could be replaced by non-native salmonids in BCG levels 3 and 4. As with the invertebrates, there was increasing abundance and dominance of tolerant species, both native and non-native, in the poorer condition levels (BCG levels 4 and 5). No BCG level 6 sites were observed in the cold and cool data set. Panelists identified level 5 rules (governing the transition from level 5 to level 6) from their experience with BCG level 6 in warmwater streams.

Table 14. Fish taxa: Ranges of attribute metrics in cold-cool transitional fish samples. BCG levels by panel consensus.

Attributes	Metric	BCG Level (Panel Consensus)				
		1 (n=1)	2 (n=13)	3 (n=14)	4 (n=9)	5 (n=7)
0 General	Total Taxa	9	1–15	4–18	10–24	10–17
	Total Individuals	470	11–207	8–598	109–534	102–1483
II Highly sensitive taxa	# Taxa	2	0–2	0–2	0–1	0
	% Taxa	22	0–100	0–25	0–7	0
	% Individuals	7	0–100	0–20	0–1	0
III Intermediate sensitive taxa	# Taxa	3	0–5	0–5	1–4	0–1
	% Taxa	33	0–67	0–36	4–22	0–10
	% Individuals	68	0–72	0–60	0–44	0–4
II + III All sensitive taxa	# Taxa	5	1–5	0–6	1–4	0–1
	% Taxa	56	33–100	0–50	4–22	0–10
	% Individuals	75	14–100	0–60	0–44	0–4
IV Intermediate tolerant taxa	# Taxa	4	0–9	1–10	4–12	3–8
	% Taxa	44	0–60	18–63	40–60	29–55
	% Individuals	25	0–83	14–88	39–83	13–93
	% Most Dom Individuals	14	0–39	8–63	18–68	7–48
V Tolerant taxa	# Taxa	0	0–1	0–5	3–8	3–7
	% Taxa	0	0–17	0–36	20–40	19–70
	% Individuals	0	0–13	0–20	4–30	1–43
	% Most Dom Individuals	0	0–13	0–16	2–18	1–19
Va Highly tolerant native taxa	# Taxa	0	0–1	0–2	0–3	0–5
	% Taxa	0	0–11	0–13	0–13	0–36
	% Individuals	0	0–1	0–1	0–18	0–85
	% Most Dom Individuals	0	0–1	0–1	0–18	0–56
VI Non-native or intentionally introduced taxa	# Taxa	0	0–1	0–3	0–1	0–1
	% Taxa	0	0–20	0–43	0–6	0–9
	% Individuals	0	0–25	0–41	0–7	0–2
	% Most Dom Individuals	0	0–25	0–41	0–7	0–2
VIa Highly tolerant non-native taxa	# Taxa	0	0	0	0–1	0–1
	% Taxa	0	0	0	0–4	0–6
	% Individuals	0	0	0	0	0–3
	% Most Dom Individuals	0	0	0	0	0–3

BCG Rule Development

For the Upper Midwest, BCG quantitative rule development can be followed by comparing Table 12 (narrative rules), Table 13 (metric distributions), and Table 15 (quantitative rules). In Table 12, the narrative rule for BCG level 2, macroinvertebrate taxa richness is: "Taxa richness is moderate to high" (Table 12). In Table 13, total taxa in BCG level 2 sites ranged from 20 to 63 invertebrate taxa (Table 13), so 20–63 is "moderate to high." The rule for total taxa (Figure 17, BCG level 2, Coolwater) was set at a midpoint of ≥ 20 taxa, with the fuzzy boundaries defined as 16 to 24. The fuzzy boundary of 16–24 defines the lower end of the "moderate" membership function for total taxa in Figure 17; membership functions were assumed to be described by straight-line segments (Figure 17). For the total taxa rule, a site with 20 invertebrate taxa would then have a membership of 50% in BCG level 2; a site with 16 taxa would have a membership of 0 (zero), and a site with 18 taxa would have a membership of 25%. A site with 24 or more taxa would have full (100%) membership in BCG level 2 for the total taxa rule. Note that the total taxa rule is the same for BCG levels 2 and 3; these BCG levels cannot be distinguished based on total taxa. Other rules must be used.

The panel's discrimination between levels 2 and 3 was primarily from richness and abundance of sensitive taxa. Attribute II taxa were always present in BCG level 2, but they were allowed to be absent in BCG level 3 (Table 13). The rules for level 2 required highly sensitive taxa (attribute II) to make up more than 5% of taxon richness and 8% of the individuals, while in level 3 the attribute II taxa were only required to be present (e.g., one taxon, one individual; Table 15). Similarly, total sensitive taxa (sum of attributes II and III) were required to comprise 30% or more of both richness and abundance in BCG level 2, but only 20% of richness, and 10% of abundance in BCG level 3. Here the panel also allowed an exception or alternative in the rules: if sensitive attribute III individuals were particularly abundant (> 40% of the community), then attribute II taxa were allowed to be absent (Alternate rule in Table 15).

The quantitative rules of Table 15 and Table 16 were developed in the same way: panel members expressed why decisions were made, with statements of what they would require to rate a higher BCG level, or what would be lost for them to rate the sample lower. These statements were later compared to the distributions of the metrics in the panel's assessed sites to yield first-iteration quantitative rules and model. The panel would then review the quantitative rules and their assessments and make adjustments to the rules (or assessments) as needed. The final quantitative rules typically emerge after two or three iterations.

Decision rules follow the patterns observed in the distributions of the metrics among BCG levels assigned by the panel. BCG level 2 requires a strong presence of sensitive (attribute II and III) taxa and, for invertebrates, sensitive EPT taxa. Other level 2 rules include minimum numbers of total taxa for invertebrates, maximum number of total taxa for fish, and low dominance of tolerant taxa in both assemblages. It is important here to emphasize that whenever absolute values are used, the sampling effort should be specified.

BCG level 3 decision rules allow slight reductions in sensitive taxa and individuals and increases in tolerant taxa. Total number of taxa requirements are the same as BCG level 2. Since metrics do not decline in lockstep with each other, the panels occasionally allowed alternative rules where an exceptionally good value in one metric could be balanced by a poor value of another. Typically, these were tradeoffs of number of sensitive taxa for number of sensitive individuals. For example, in the invertebrate rules (Table 15), the percent sensitive (attributes I, II, and III) taxa and individuals—were subject to alternate rules: If the value of the percent sensitive taxa metric is > 20%, then the percent sensitive individuals must be > 10%. Alternatively, if the value of the percent sensitive taxa metric is > 40%, then the percent sensitive individuals metric need only be > 5%.

BCG level 4 is characterized by decreased richness and abundance of sensitive taxa. However, sensitive taxa must still be present above a minimum floor. The disappearance of sensitive taxa is what typically discriminates level 5 from level 4, as well as an increase in the percent tolerant (attribute V) individuals (Table 12, Table 16, Table 17).

Table 15. Benthic macroinvertebrate taxa: Decision rules for macroinvertebrate assemblages in coldwater and coolwater (transitional cold-cool) streams; samples with > 200 organisms. Rules show the midpoints of fuzzy decision levels, followed by the range of the membership function. The midpoint is where membership in the given BCG level is 50% for that metric.

BCG Level	Metrics	Coldwater		Coolwater	
		Rule		Rule	
2	# Total taxa	≥ 14 (11–16)		≥ 20 (16–24)	
	% Most sensitive taxa (Att I + II)	> 10% (7%–13%)		> 5% (3%–7%)	
	% Most sensitive individuals (Att I & II)	—		> 8% (6%–10%)	
	% Sensitive taxa (Att II + III)	> 30% (25%–35%)		> 30% (25%–35%	
	% Sensitive individuals (Att II + III)	> 30% (25%–35%)		> 30% (25%–35%)	
	% Most dominant tolerant taxa (Att V)	< 5% (3%–7%)		—	
	% Sensitive EPT taxa (Att I + II + III)	> 10% (7%–13%)		> 10% (7%–13%)	
		Rule	Alt Rule	Rule	Alt Rule
3	# Total taxa	≥ 14 (11–16)		≥ 20 (16–24)	
	# Most sensitive (Att I + II) taxa	—	present	NA	
	% Sensitive taxa (Att II + III)	> 20% (15%–25%)	> 40% (35%–45%)	> 20% (15%–25%)	
	% Sensitive individuals (Att II + III)	> 10% (7%–13%)	> 5% (3%–7%)	> 10% (7%–13%)	> 40% (35%–45%)
	% Most dominant intermediate tolerant taxa (Att IV)	< 50% (45%–55%)		—	
	% Tolerant (Att V) individuals	< 20% (15%–25%)		—	
	% Most dominant tolerant taxa (Att V)	—		< 10% (7%–13%)	
	% Sensitive EPT taxa (Att I + II + III)	> 10% (7%–13%)		> 10% (7%–13%)	
		Rule		Rule	
4	# Total taxa	≥ 8 (6–10)		≥ 14 (11–16)	
	% Sensitive taxa (Att II + III)	> 10% (7%–13%)		> 10% (7%–13%)	
	% Sensitive individuals (Att II + III)	> 5% (3%–7%)		> 6% (4%–8%)	
	% Tolerant (Att V) individuals	< 40% (35%–45%)		< 60% (55%–65%)	
	Number of sensitive EPT taxa (Att I + II + III)	present		present	
		Rule		Rule	
5	# Total taxa	≥ 8 (6–10)		≥ 14 (11–16)	
	% Tolerant (Att V) individuals	< 60% (55%–65%)		—	
	% Most dominant tolerant taxa (Att V)	—		< 60% (55%–65%)	

Table 16. Fish taxa: Decision rules for fish assemblages in coldwater and coolwater (cold-cool transitional) streams. Rules show the midpoints of fuzzy decision levels, where membership in the given BCG level is 50% for that metric.

<table>
<tr><th rowspan="2">BCG Level</th><th rowspan="2">Metrics</th><th colspan="4">Coldwater</th><th colspan="2">Coolwater</th></tr>
<tr><th colspan="2">Brook Trout (BT) Native</th><th colspan="2">BT Non-native</th><th>BT Native</th><th>BT Non-native</th></tr>
<tr><td rowspan="8">1</td><td></td><td colspan="4"></td><td colspan="2">Meets Coldwater level 1,
OR Coolwater rules below:</td></tr>
<tr><td># Total taxa</td><td colspan="4">≤4 (2–5)</td><td colspan="2">> 3 and < 14 (2–5 and 11–16)</td></tr>
<tr><td>% Most sensitive taxa (Att II)</td><td colspan="4">Present</td><td colspan="2">Present</td></tr>
<tr><td>% Brook trout individuals</td><td colspan="2">Present</td><td colspan="2">Absent</td><td>Present</td><td>Absent</td></tr>
<tr><td>% Sensitive taxa (Att II + III)</td><td colspan="4">> 50% (45%–55%)</td><td colspan="2">> 40% (35%–45%)</td></tr>
<tr><td>% Sensitive individuals (Att II + III)</td><td colspan="4">> 60% (55%–65%)</td><td colspan="2">> 40% (35%–45%)</td></tr>
<tr><td>% Tolerant (Att V + Va + VIa) individuals</td><td colspan="4">< 5% (3%–7%)</td><td colspan="2">< 5% (3%–7%)</td></tr>
<tr><td>% Non-native salmonids (Att VI)</td><td colspan="4">Absent</td><td colspan="2">Absent</td></tr>
<tr><td rowspan="9">2</td><th rowspan="2">Metrics</th><th colspan="2">BT Native</th><th colspan="2">BT Non-native</th><th rowspan="2">BT Native</th><th rowspan="2">BT Non-native</th></tr>
<tr><th>Alt 1</th><th>Alt 2</th><th>Alt 1</th><th>Alt 2</th></tr>
<tr><td># Total taxa</td><td colspan="4">If watershed size ≤ 10 mi^2, < 8 (6–10)
If watershed size > 10 mi^2, > 3 and < 14 (2–4 and 11–16)</td><td colspan="2">< 20 (16–24)</td></tr>
<tr><td>% Most sensitive taxa (Att II)</td><td colspan="2">Present</td><td colspan="2">NA</td><td>Present</td><td>NA</td></tr>
<tr><td>% Brook trout individuals</td><td colspan="2">Present</td><td colspan="2">NA</td><td>Present</td><td>NA</td></tr>
<tr><td>% Sensitive taxa (Att II + III)</td><td>> 40% (35%–45%)</td><td>> 20% (15%–25%)</td><td colspan="2">> 20% (15%–25%)</td><td colspan="2">> 30% (35%–45%)</td></tr>
<tr><td>% Sensitive individuals (Att II + III)</td><td colspan="2">NA</td><td>> 70% (65%–75%)</td><td>NA</td><td colspan="2">> 12% (9%–15%)</td></tr>
<tr><td>% Brook trout: total salmonid individuals</td><td colspan="2">> 40% (35%–45%)</td><td colspan="2">NA</td><td>> 40% (35%–45%)</td><td>NA</td></tr>
<tr><td>% Tolerant non-salmonid (Att V + Va + VIa) individuals</td><td>< 10% (7%–13%)</td><td>Absent</td><td>NA</td><td>< 10% (7%–13%)</td><td colspan="2">< 20% (15%–25%)</td></tr>
</table>

BCG Level	Metrics	Coldwater Rule	Coldwater Alt Rule	Coolwater Rule	Coolwater Alt Rule
		(brook trout native/non-native status not used)			
3	# Total taxa	If watershed size > 10 mi^2, > 5 (3–7)		< 20 (16–24)	
	% Salmonid individuals	Present		–	
	% Sensitive & non-native salmonid (Att I + II + III + VI) taxa	> 25% (20%–30%)		–	
	% Sensitive & non-native salmonid (Att I + II + III + VI) individuals	> 20% (15%–25%)		–	
	% Non-native salmonid (Att VI): total sensitive (Att I + II + III + VI) individuals	< 70% (65%–75%)		–	
	% Sensitive taxa (Att II + III)	–		≥ % Tolerant (Att V + Va + VIa) taxa	NA
	% Sensitive individuals (Att II + III)	–		NA	≥ 2*Tolerant (Att V + Va + VIa) % individs
	% Most dominant intermediate tolerant taxa (Att IV)	–		If watershed size > 10 mi^2, < 40% (35%–45%)	
	% Extra tolerant individuals (Att Va + VIa)	–		< 5% (3%–7%)	
4	**Metrics**	**(no alternate rules)**			
	% Sensitive & salmonid taxa (Att II + III + VI)	> 5% (3%–7%)		> 5% (3%–7%)	
	% Sensitive & salmonid individuals (Att II + III + VI)	> 5% (3%–7%)		> 5% (3%–7%)	
	% Tolerant taxa (Att V + Va + VIa)	< 45% (40%–50%)		–	
	% Extra tolerant individuals (Att Va + VIa)	< 10% (7%–13%)		< 20% (15%–25%	
5	**Metrics**	**(no alternate rules)**			
	# Total taxa	> 2 (1–3)		> 3 (2–4)	
	% Intermediate tolerant taxa (Att IV)	> 10% (7%–13%)		> 10% (7%–13%)	

Model Performance

In general, the fuzzy model identified 75%–80% of samples as primarily a single BCG level (75% membership or greater). Approximately 10%–15% of samples had a large minority membership in an adjacent BCG level to the "nominal" level (25%–40% membership), and approximately 10%–15% of assessments are ruled ties or near-ties between adjacent BCG levels (minority membership > 40%).

To measure model performance with the calibration data sets, two matches in BCG level choice were considered: an exact match, where the BCG decision model's nominal level matched the panel's majority choice; and a "minority match," where the model predicted a BCG level within one level of the majority expert opinion. When model performance was evaluated in this calibration data set, the coldwater macroinvertebrate model matched exactly with the regional biologists' BCG level assignments on 97.6% of the coldwater samples (Table 17). In the single sample without agreement, the model assignment was one level better than the majority expert opinion.

In order to confirm the model, panelists made BCG level assignments on additional samples. When nominal level assignments from the BCG decision model were compared to the panelists' nominal level assignments in the confirmation data set, the model matched exactly with the regional biologists' BCG level assignments on 80% or more of the samples (Table 17). In both cold and coolwater, three confirmation samples were rated differently by model and panel, where the model rated the samples as being one BCG level better than the majority expert opinion. Based on the combined results, in 89% of cases, the macroinvertebrate model predicts the same BCG level as the majority expert opinion.

Table 17. Benthic macroinvertebrate and fish taxa: Model performance—cold and coolwater samples

	Benthic macroinvertebrates				Fish			
Model	**Coldwater**		**Cool-transitional**		**Coldwater**		**Cool-transitional**	
Difference	**Calib.**	**Conf.**	**Calib.**	**Conf.**	**Calib.**	**Conf.**	**Calib.**	**Conf.**
2 better	0	0	0	0	0	0	0	1
1 better	2	3	1	3	3	3	3	5
same	39	13	31	15	47	21	38	17
1 worse	1	0	2	0	2	1	1	2
2 worse	0	0	3	0	0	0	0	0
Total # Samples	**42**	**16**	**34**	**18**	**52**	**25**	**42**	**25**
% Correct	**98%**	**81%**	**91%**	**83%**	**90.4%**	**84%**	**90%**	**68%**

4.1.2.4 Example #2: Quantitative Rules and Decision System for Diatoms, New Jersey

New Jersey DEP developed and calibrated a BCG model for sampled diatoms in northern New Jersey streams (Gerritsen et al. 2014). The models were developed using data collected by the Academy of Natural Sciences for New Jersey DEP. Workshop participants included scientists from around the United States. The calibrated BCG models will allow New Jersey to express and assess goals for classes of water bodies in terms of their biological condition.

Study sites

The data set consisted of 42 samples collected from streams and rivers in northern New Jersey. Sites were located in the Northern Piedmont (25), the Northern Highlands (6), the Ridge and Valley (7), the Atlantic Coastal Pine Barrens (3), and the Middle Atlantic Coastal Plain (1) ecoregions (Omernik 1987; Woods et al. 2007). Land-use in the Piedmont is primarily urban and agriculture, whereas in the Highlands and the Ridge and Valley it is predominantly forest and agriculture (USEPA 2000a). Within ecoregions, the study sites had relatively similar natural environmental conditions (e.g., geology, geomorphology), but with a wide range of nutrient concentrations.

A narrative description was derived from discussions with the panelists about why individual sites were assessed at a certain level (Table 18). The rules were calibrated from the narrative description and the 30 calibration samples rated by the group, and the rules were adjusted so that the model would replicate the panel's decisions as closely as possible. Panel members were highly quantitative in their thinking and deliberations, and they developed the first iteration of quantitative rules based on the narrative descriptions.

<u>Rule Development</u>

Rules adopted for the quantitative decision model are listed in Table 19. BCG level 1 has five rules: one on taxa richness, two rules on abundance of sensitive taxa, and two rules on abundance of tolerant taxa. For BCG level 1, sensitive taxa are required to be dominant, and tolerant taxa are very minor constituents of the community. The rules for BCG level 2 are similar to level 1, but all have been relaxed to some extent. The largest relative difference between levels 1 and 2 is that attribute II individuals are required to be highly abundant in level 1 (roughly 35% or more), but they are subdominant in level 2 (10% or more).

In BCG level 4, sensitive individuals are greatly diminished, but still present (9% or more), and tolerant taxa can occur at higher abundances. There are only three rules for BCG level 5: tolerant taxa may not exceed 40% of taxa or 80% of individuals. Samples that fail to meet the BCG level 5 requirements would be assigned to BCG level 6, but no such samples were encountered in this data set.

<u>Model Performance</u>

To evaluate the performance of the 40-sample calibration data set and the 10-sample confirmation data set, the number of samples where the BCG decision model's nominal level exactly matched the panel's majority choice ("exact match"), and the number of samples where the model predicted a BCG level that differed from the majority expert opinion ("anomalous" samples) were assessed. Then, for the anomalous samples, the degree of differences among the BCG level assignments, and also whether there was a bias was examined (e.g., did the BCG model consistently rate samples better or worse than the panelists?).

Two types of ties were taken into account: (1) BCG model ties, where there is nearly equal membership in two BCG levels (e.g., membership of 0.5 in BCG level 2 and membership of 0.5 in BCG level 3); and (2) panelist ties, where the difference between counts of panelist primary and secondary calls is less than or equal to 1 (e.g., 4–4 or 4–3 decisions). If the BCG model assigned a tie, and that tie did not match with the panelist consensus, it was considered to be a difference of half a BCG level (e.g., if the BCG model assignment was a BCG level 2/3 tie and panelist consensus was a BCG level 2, the model was considered to be "off" by a half BCG level; or more specifically, the model rating was a half BCG level worse than the panelists' consensus). The BCG model was also considered to differ by a half level if the panelists assigned a tie and the BCG model did not.

Results show that the diatom BCG model performed well (Table 20). The models assigned scores that are within a half BCG level or better on 100% of the samples in both the calibration and confirmation data sets (Table 18). When half levels were considered, the BCG model rated three of the calibration samples a half level worse than the panelists, and five confirmation samples (two better, three worse). Based on results from the calibration data set, the model has a slight bias towards rating samples a half level worse than the panel consensus.

Table 18. Narrative description of diatom assemblages in six BCG levels for streams of northern New Jersey. Definitions are modified after Davies and Jackson (2006).

BCG level 1	**Definition:** Natural or native condition—*native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability*
	Narrative: BCG level 1 streams in northern New Jersey highlands are oligotrophic, with a mature forested watershed. Unlike macroinvertebrates, the diatom community is relatively depauperate, with typically 15–20 taxa in a 500-count sample. The top dominant taxa are extreme low-nutrient adapted taxa of attributes II and III (e.g., *Achnanthes subhudsonis* or *Achnanthidium rivulare)*. Subdominants (up to 10% abundance) may include attribute IV taxa. Tolerant taxa (attribute V) make up a very small fraction of the community.
BCG level 2	**Definition:** Minimal changes in structure of the biotic community and minimal changes in ecosystem function—*virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability*
	Narrative: BCG level 2 streams are very similar to level 1, however, a slight increase in disturbance or enrichment has allowed more diatom taxa to colonize (20–40 total taxa). Richness is slightly higher than level 1, but low nutrient taxa (attribute II and III) are dominant. There may be several tolerant taxa, but their abundance is low.
BCG level 3	**Definition:** Evident changes in structure of the biotic community and minimal changes in ecosystem function—*Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but intermediate sensitive taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system*
	Narrative: Richness is higher than level 2 (> 30 taxa). Dominant taxon may or may not be sensitive (attribute II or III). Tolerant taxa have increased to more than 10% of the assemblage, and some of the tolerant taxa are now in the subdominant category.
BCG level 4	**Definition:** Moderate changes in structure of the biotic community and minimal changes in ecosystem function—*Moderate changes in structure due to replacement of some intermediate sensitive taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes*
	Narrative: BCG level 4 sites tend to have the highest taxa richness as more diatom niches open up with increased enrichment, light penetration (from canopy loss), and moderate sedimentation. Sensitive species and individuals are still present but in reduced numbers. The persistence of some sensitive species indicates that the original ecosystem function is still maintained albeit at a reduced level. Intermediate and tolerant taxa may be dominant, sensitive taxa are often still subdominant.
BCG level 5	**Definition:** Major changes in structure of the biotic community and moderate changes in ecosystem function—*Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials*
	Narrative: Overall diversity is still high, but may be slightly reduced from level 4. Sensitive species may be present but their functional role is negligible within the system. The most abundant and dominant taxa are tolerant or have intermediate tolerance, and there may be relatively high diversity within the tolerant organisms.
BCG level 6	**Definition:** Major changes in structure of the biotic community and moderate changes in ecosystem function—*Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials*
	Narrative: Heavily degraded from urbanization and/or industrialization. No level 6 samples were encountered by the panel.

Table 19. BCG quantitative decision rules for diatom assemblages in northern New Jersey streams. The numbers in parentheses represent the lower and upper bounds of the fuzzy sets. BCG level 6 is not shown, because there are no specific rules for level 6: If a site fails level 5, it falls to level 6. Shaded rules under BCG level 3 are alternate rules, that is, at least one must be true for a site sample to meet BCG level 3.

Attribute metric	Threshold
BCG Level 1	
# Total taxa	≤ 20 (15–25)
% Attribute II+III individuals	≥ 65% (60%–70%)
% Attribute II individuals > % Attribute III individuals; expressed as (% Att II–% Att III)	> 0% (-10% to 10%)
% Attribute V+VI individuals	< 2.5% (1%–4%)
% Most dominant Attribute V or VI taxon	≤ 1% (0%–2%)
BCG Level 2	
# Total taxa	≤ 40 (35–45)
% Attribute II individuals	≥ 10% (5%–15%)
% Attribute II+III individuals	≥ 50% (45%–55%)
% Attribute II+III taxa	≥ 15% (10%–20%)
% Attribute V+VI individuals	≤ 10% (5%–15%)
% Most dominant Attribute V or VI taxon	≤ 5% (3%–7%)
BCG Level 3	
# Attribute II+III taxa	≥ 5 (2–8)
# Attribute II taxa	≥ 1 (0–1)
Most dominant taxon*	Att II or 3
Alt 1: % Attribute II+III taxa	≥ 15% (10%–20%)
Alt II: % Attribute II+III individuals	≥ 15% (10%–20%)
% Attribute V+VI individuals	≤ 30% (25%–35%)
% Most dominant Attribute V or VI taxon	≤ 10% (5%–15%)
BCG Level 4	
% Attribute II+III individuals	≥ 9% (5%–13%)
% Attribute V+VI individuals	≤ 65% (60%–70%)
% Most dominant Attribute V or VI taxon	≤ 40% (35%–45%)
BCG Level 5	
% Attribute V+VI taxa	≤ 40% (35%–45%)
% Attribute V+VI individuals	≤ 80% (75%–85%)

* Dominant taxon must be sensitive (Att II or III); membership = 0 if rule fails

Table 20. Model performance for calibration and confirmation samples. "½ better" indicates models scored the sample ½ BCG level higher than the panel; e.g., Panel score was 4 and model score was 3–4 tie. Half-level mismatches are counted half the value of full matches. No mismatches exceeded ½ BCG level.

Difference (model vs. panel consensus call)	Calibration		Confirmation	
	Number	Percent	Number	Percent
model 1 level better	0	0	0	0
model ½ level better	0	0	2	17
exact match	27	90	7	58
model 1/2 level worse	3	10	3	25
model 1 level worse	0	0	0	0
Total # Samples	30	95	12	79

4.2 Calibrating Indices to the Biological Condition Gradient

Most states have developed biological indices for their streams and wadeable rivers (USEPA 2002). In the initial development of BCGs, common questions asked by states included:

- What is the relationship between the BCG and the state's existing biological index, or indices?
- Does the BCG replace the existing biological index, or indices?
- How can the BCG and the existing biological index, or indices, be used together to better assess ALUs?

The linkage between a biological index and the BCG could be addressed in a state program review (USEPA 2013a) and/or as a topic of discussion within the expert panel. Existing indices could be evaluated for how extensively they include attributes of the BCG or how the BCG decision criteria match up with the metrics that comprise the index. If needed, recommendations for specific technical improvements and analyses can then be made to guide the redevelopment of an index and/or refine the BCG model.

As in section 4.1., the objective is to calibrate a BCG model with a quantitative model, or in this case, an index that will duplicate the expert panel BCG assessments for new samples and water bodies, without having to reconvene the panel. In this approach, a conventional IBI (e.g., Karr 1986) or predictive biological index model (e.g., Hawkins et al. 2000b; Wright 2000) could be calibrated to the expert-derived BCG. While the seminal works about these indices preceded the BCG, they are based on parallel ecological concepts, and to varying degrees each incorporates BCG attributes. As an example of this, Table 21 illustrates the overlap between the 10 BCG attributes and a selection of fish and macroinvertebrate indices for freshwater streams and wadeable rivers. For the fish indices, the metrics used for each capture the more commonly measured attributes I–VI (taxa composition and effects of non-native taxa), but they also address attributes VII (organism condition), VIII (ecosystem function), and X (ecosystem connectance). The routine inclusion of the deformities, erosions, lesions, and tumors (DELT) anomalies metric (e.g., measure of deformities, erosion, lesions, and tumors) in all fish indices contains attribute VI. Functional feeding and reproduction guilds that are routinely included in fish indices might provide a surrogate for attribute VIII. The inclusion of diadromous metrics provides for the direct inclusion of species that depend on access to and from coastal rivers for completing their life cycles. Other metrics that include species that are dependent on free access to a drainage network can illustrate the concept of connectivity in inland streams and rivers. Attribute IX (spatial and temporal extent of detrimental effects) can be accounted for by the spatial extent of the sampling design and is independent of the composition of fish IBIs. For the macroinvertebrate metrics in Table 21, coverage of attributes I–V is provided by most biological indices used by states. It is also possible to develop non-native taxa metrics for attribute VI (presence and effect of non-native taxa) and metrics for attribute X (ecosystem connectance). Biological metrics could serve as a surrogate for attribute X—Unionid mussels might be a good choice given their dependency on fish hosts for dispersal and to sustain their populations. The key point is that (MMIs) have been developed from the same or parallel concepts as the BCG.

Table 21. Cross referencing the 10 BCG attributes with selected fish IBI and macroinvertebrate MMI metrics for streams and wadeable rivers

BCG Attribute	Fish IBI Metrics	Macroinvertebrate Metrics
I. Historically documented, sensitive, long-lived, or regionally endemic taxa	Great River species Sensitive sucker species Native salmonid species American eel numbers & size classes Selected diadromous species	Unionid mussels # of *Pteronarcys* species
II. Highly sensitive taxa	Highly intolerant species Sensitive species Temperate stenotherms Native salmonids	Mayfly & EPT metrics
III. Intermediate sensitive taxa	Moderately intolerant species sensitive species Round-bodied suckers	Mayfly, caddisfly, Tanytarsini midge, EPT metrics
IV. Intermediate tolerant taxa	Included in native species richness Number of minnow species Number of sunfish species	Taxa richness, caddisfly, Dipteran taxa, Non-insect & Other Dipteran taxa
V. Tolerant taxa	Highly tolerant species	Tolerant taxa % Abundance tolerant Taxa
VI. Non-native or intentionally introduced species	Exotic and introduced species of intracontinental origin Non-native species	%*Corbicula; Dreissenid mussels*
VII. Organism condition	DELT anomalies Total native species biomass	Head capsule deformities
VIII. Ecosystem function	Proportion in functional feeding groups Specialist metrics, i.e., fluvial specialists & dependents	%Other Dipteran & non-insects %Filterers %Grazers/scrapers %Clingers
IX. Spatial and temporal extent of detrimental effects	Accounted for in spatial sampling design	(Same as fish)
X. Ecosystem connectance	Diadromous species Native Salmonids Non-indigenous species	Unionid mussel

Indices that are currently in widespread use are of two basic types:

- Indices comprised of metrics that are the aggregation of species/taxa abundance data based on taxonomy, environmental tolerance, functional role, assemblage condition, and organism condition. Each metric is calibrated on a range from best to poorest conditions and also with respect to natural factors such as watershed size. The index development process usually includes an examination of tens to hundreds of candidate metrics and reducing this list to the most relevant and/or responsive 8–12 metrics (approximately). The metrics can be somewhat independent in response to each other and, when summed together, can either dilute or amplify an interpretation. They are useful in observing trajectory, but they may require recalibration to the BCG attributes before they can produce a BCG assessment. Most of this class of indices have been developed for fish, macroinvertebrates, and algae although development for other groups such as Unionid mussels have been attempted (Barbour et al. 1999). Within this broad class of indices are the classic IBIs that follow the seminal guidance of Karr et al. (1986), most of which have been developed for fish assemblages, but some for macroinvertebrates. While the original IBI was developed for central Illinois fish assemblages, Karr et al. (1986) provided guidelines about the possible application to other regions and other aquatic assemblages. This was done knowing that different metrics would be needed, but the goal was to maintain the essential attributes and ecological content of an IBI. Other multimetric approaches have been developed and applied for macroinvertebrates that, while utilizing a generally similar process, are somewhat distinctive from IBIs in having metrics that are predominantly based on taxa attributes (Plafkin et al. 1989; Barbour et al. 1999).
- Predictive models, where the observed species composition at a site is compared to an idealized reference site predicted from a multivariate statistical model. These models develop an expected taxon list and use the O/E ratio (e.g., the River InVertebrate Prediction and Classification System, RIVPACS, e.g., Wright (2000) and the AUStralian RIVer Assessment System, AUSRIVAS, Simpson and Norris (2000)). A second approach has been to use a multivariate similarity index between a specific sample and a centroid defined by undisturbed reference sites (e.g., Percent Model Affinity, Novak and Bode 1992; BEAST, Reynoldson et al. 1995; dissimilarity, Van Sickle 2008). Predictive approaches have also been applied in a multimetric framework, in which expectations for the metrics are based on environmental variables (Chen et al. 2014; Esselman et al. 2013; Moya et al. 2011; Oberdorff et al. 2002; Pont et al. 2006; Pont et al. 2009).

Ideally, the calibration of MMIs are based on minimally disturbed reference sites and with respect to natural classification strata such as bioregions, thermal gradients, and other factors that determine the baseline expectations of a regional aquatic fauna (Stoddard et al. 2006). Some have used all the data assuming that the best, or least disturbed, sites reflect the highest possible condition (Blocksom 2003; Stoddard et al. 2006). Such an assumption should be evaluated by expert opinion before it is accepted that the best condition found in a data set reasonably represents the highest expected condition. Calibration techniques have also evolved from the ordinal approach of Fausch et al. (1984) to continuous calibration techniques (Blocksom 2003; Mebane et al. 2003) that could be applied to BCG development. The expectations for achieving a high level of rigor in this process are described in EPA's Biological Assessment Program Review document (USEPA 2013a). As such, the level of technical rigor achieved in these important calibration steps can also affect the ability to measure condition along the BCG.

As with the development of the BCG, it is also necessary to test an index or model across a gradient of different environmental stressors. The ability to quantify departures from reference-derived thresholds is an important step in evaluating any assessment model.

4.2.1 Biological Condition Gradient Thresholds for Multimetric Indices and Multivariate Models

Indices and models as generally described herein should accurately translate to a position along the BCG. However, the proficiency of a particular index or model to actually accomplish this, at a particular level of resolution, is dependent on the level of detail and rigor applied in construction of the index or model and the calibrated BCG model. EPA (2013a) provides a standardized way to evaluate the technical strengths and gaps in a biological assessment program and to determine how well a particular biological assessment protocol discriminates incremental changes in biological condition (i.e., the higher the level of rigor, the more precision is achieved in incremental measurement along a gradient of stress).

However, simply stratifying an index scoring range along the BCG is neither sufficient nor recommended, especially if an index has not been explicitly developed within the conceptual framework of the BCG or the BCG attributes have not be reconciled with the metrics that comprise the index. For example, metrics in a MMI may have been selected because of strong known response to current or selected stressors and may not comprehensively characterize the full range of biological conditions, while the BCG decision rules are based on benchmarks for undisturbed or minimally disturbed conditions. This has been a challenge, especially with the upper BCG levels where reference analogs to BCG levels 1, 2, 3, or sometimes even 4 either do not exist or have not been identified. If this is the case, it will be necessary to revisit the existing index derivation and BCG model calibration and possibly revise either one, or both, for better correspondence. This task can be accomplished by the state biological assessment and criteria program, but it should be done in collaboration with the full expert panel that developed the BCG model and the underlying quantitative decision rules. As described in Chapter 3, through an iterative process, scoring criteria can be developed for new or refined indices that correspond with biologists' consensus about narrative descriptions of the levels in the BCG.

4.2.1.1 Calibrating Index Scores: Connecticut Stream Example

The set of sites that have been assigned to levels of the BCG are used to calibrate index scores. Index scores for the sites are examined, and, if separation of the index scores among levels is good, then index thresholds can be selected to maximize the ability to discriminate among the levels. This is demonstrated in the Connecticut case example below and by the Minnesota case study where IBI thresholds for refined ALUs were based on the correspondence between their IBIs and BCG levels (section 6.4). In the Connecticut example, BCG calibration and a macroinvertebrate MMI were developed at the same time. The MMI consisted of seven metrics (Table 22; Gerritsen and Jessup 2007b), including an abundance-weighted average of BCG attributes II through VI.

Table 22. Correlations (Pearson r) among Connecticut MMI index metrics

#	Metric	1	2	3	4	5	6	7
1	Ephemeroptera taxa (adj.)	•						
2	Plecoptera taxa	0.58	•					
3	Trichoptera taxa	0.57	0.50	•				
4	% sensitive EPT (adj.)	0.69	0.54	0.52	•			
5	Scraper taxa	0.67	0.50	0.75	0.52	•		
6	BCG Taxa Biotic Index	-0.76	-0.76	-0.68	-0.74	-0.69	•	
7	% dominant genus	-0.61	-0.54	-0.62	-0.59	-0.60	0.66	•

Note: Adj. = Metric scoring was adjusted for catchment size.

The Connecticut stream MMI uses metrics that are similar in objective to the BCG attributes, but which are calculated somewhat differently (e.g., EPT taxa metrics in the MMI include taxa considered to be attributes II, III, IV; and attribute II includes taxa from the EPT orders, as well as a few dipteran and beetle taxa). The total MMI score is based on the average of all metrics, while BCG decisions are based on decision-specific critical attributes (e.g., attributes II and III for the higher levels and attribute V for lower levels). Concordance of the two assessment endpoints is strong (Figure 20). Figure 20 shows the predicted results of the BCG inference model.

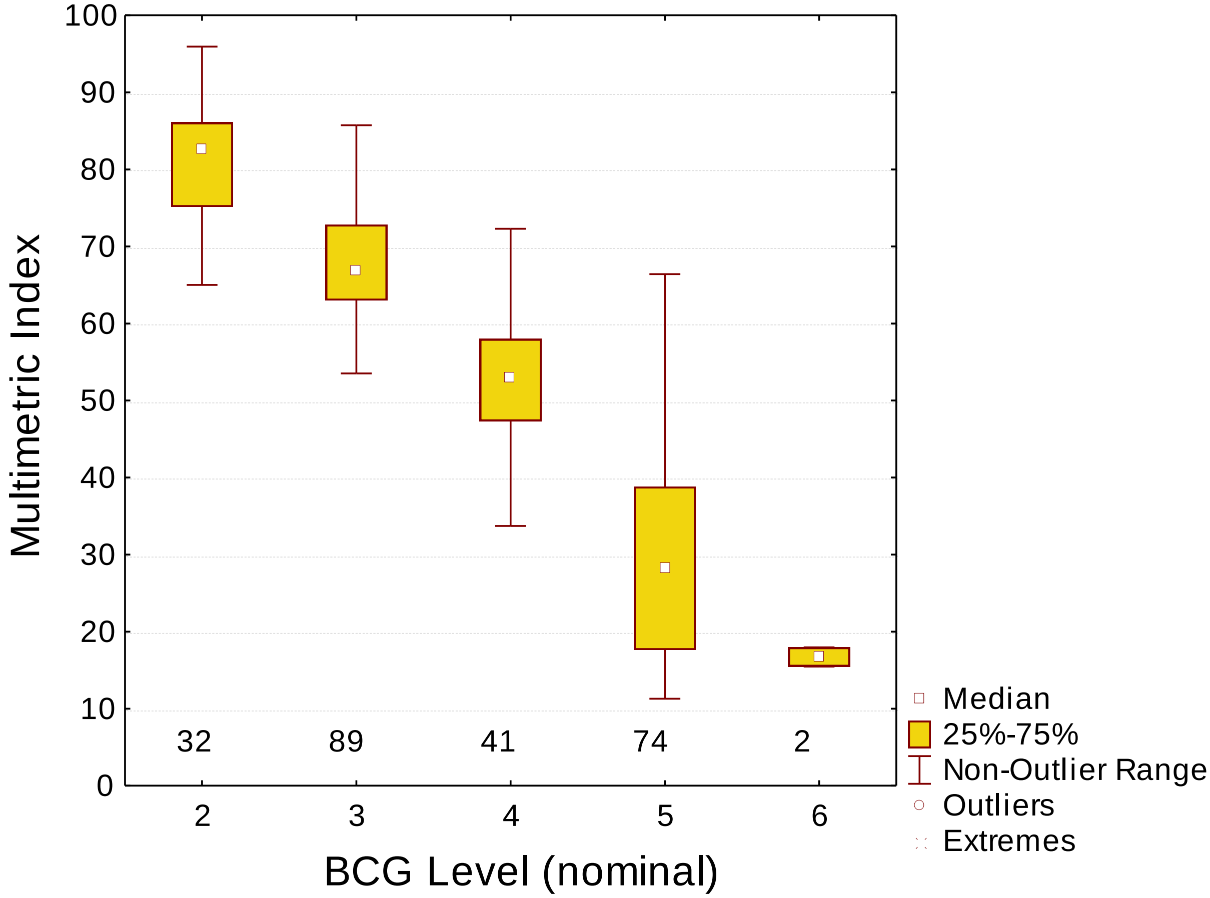

Figure 20. Connecticut MMI by BCG levels, estimated from decision analysis model. Number of samples given below boxes.

In spite of these differences, MMI scores could be used to separate levels (Figure 20). Potential MMI scoring thresholds are given in Table 23.

Table 23. Scoring thresholds for the Connecticut MMI to correspond to BCG levels

BCG Level	MMI Scoring Range
Levels 1, 2	> 75
Level 3	60–74.9
Level 4	43–59.9
Level 5	20–42.9
Level 6	> 20

The BCG decision model and the MMI were in overall concordance on the assessments from the two methods. The scoring range of the MMI was broken into categories corresponding to BCG levels. This resulted in disagreement of 32% of multimetric scores compared to the BCG decision model, but disagreements were never by more than a single level. There was no bias in the direction of disagreement among models, determined by the similar number of MMI assessments that were better or worse than the corresponding BCG assessments.

An additional example of an approach to reconcile an existing index to the BCG is included in Appendix B. This example involves an innovative technique to "back calculate" a historically representative IBI (Appendix B1). In this case it helped to clarify the position of an IBI based on current-day stressors for the Upper Mississippi River.

4.3 Statistical Models to Predict Expert Decisions: Multivariate Discriminant Model Approach

Another approach to quantify expert consensus and develop a BCG model is use of multivariate statistical models to predict expert judgment. For example, Maine DEP developed a set of multivariate linear discriminant models to simulate the expert consensus and predict a site assessment (Danielson et al. 2012; Davies et al. In press), and the United Kingdom Environmental Agency defined ranges of scores of two indices (their RIVPACS index and a tolerance index) that correspond to expert consensus (Hemsley-Flint 2000). Both of these approaches utilize one or more multivariate statistical models to predict the expert judgment in assessments. The following section describes Maine's use of linear discriminant models to discern levels of biological condition.

4.3.1 Approach

The objective of the discriminant model approach is the same as that of the quantitative rule development approach described in section 4.1: to develop a predictive model that will duplicate the decisions of the expert panel, so that new water bodies can be assessed without having to reconvene the panel. As with the rule development, the discriminant model (a multivariate statistical model) uses the same data available to the expert panel.

Discriminant analysis can be used to develop a model that will divide, or discriminate, observations among two or more groups whose membership characteristics have been defined *a priori* (i.e., in advance) of the construction of the model. This is accomplished through use of a model-building or "learning" data set in which samples have been assigned into the groups of interest, for example by expert consensus much like the expert panel process discussed in section 4.1.1. In short, for purposes of calibrating a BCG model, a discriminant function model can be developed from a biological data set where sites in a training data set have previously been assigned to BCG levels. A discriminant function model is a linear function combining those input variables that most successfully contribute to group definition and discrimination among groups. The resulting model yields the maximum separation (discrimination) among the groups (e.g., levels of the BCG). The analysis objectively identifies the best discriminatory variables and weights their relative contribution to the discriminatory model using coefficients. Selection of input variables is aided by initial exploratory data analysis to investigate relationships between biological response variables and physical stream characteristics (width, depth, velocity, elevation, temperature), and by data reduction techniques to eliminate highly correlated variables.

The linear discriminant model (LDM) approach may reveal subtle discriminatory variables within the data set that the biologists might not have recognized as important. This feature of statistical selection of variables contributes to building a highly discriminatory model. In construction of an LDM, input variables can also be included in the model on the basis of the judgment of experts that the variable provides an important link to assessment of the specific biological values that are stated in narrative biological criteria. Once constructed, the model can be used to objectively and consistently determine membership in a BCG level for new observations where the level is unknown. Maine uses this method to determine whether streams are meeting biological criteria for the state's tiered ALUs.

Although it requires statistical expertise to develop, another advantage of discriminant analysis is that it uses established and well-documented statistical methodology, with known confidence limits, and it reports group membership of a sample as probability statements, providing an understanding of the degree of certainty of the reported result. While LDMs require a relatively large set of assigned sites to calibrate the model (approximately 20 per group due to dependence upon having a suitable number of degrees of freedom, Manly 1991; Wilkinson 1989), accuracy of the model to the expert-assigned calibration and test sites can be as high as 89%–97%[6] (Davies et al. In press; Shelton and Blocksom 2004).

Using a discriminant model to develop biological criteria requires both a set of model-building data to develop the model and confirmation data to test the model. If a sufficient number of samples are available, the training and confirmation data may be from the same biological database, randomly divided into two sets (60% to 70% of data for calibration), or they may be drawn from two or more years of survey data. All sites in each data set are assigned to BCG levels by the expert workgroup.

Depending upon the required precision of the model, one or more discriminant function models that function in a hierarchical fashion may be developed from the model-building set to predict level membership from biological data. Building a set of nested, hierarchical models is an effective way of improving overall predictive accuracy (Davies et al. In press). Once developed, the model is applied to the confirmation data set to determine how well it can assign sites to levels using independent data not used to develop the model. More information on discriminant analysis can be found in many available textbooks on multivariate statistics (e.g., Jongman et al. 1987; Legendre and Legendre 1998; Ludwig and Reynolds 1998; Rencher 2003).

4.3.1.1 Example—Maine Discriminant Model for Benthic Macroinvertebrate Assemblages (Source: Shelton and Blocksom 2004)

Maine has four designated use classifications for its streams, AA, A, B, and C, with three corresponding ALUs. Classes AA (Maine's outstanding natural resource waters) and A correspond to BCG levels 1 and 2 (per Maine's narrative criteria, "as naturally occurs"), and they are not distinguishable based on Maine's biological assessment method. Class B ("no detrimental change") corresponds approximately to BCG level 3, and Class C ("maintain structure and function") corresponds approximately to BCG level 4.[7] Streams in poorer condition than Class C, comprising BCG levels 5 and 6, are not in attainment (NA) of minimum state ALU standards. Section 6.5 provides details of implementation and application of

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[6] Based on jack-knife tests of the combined nested LDMs in Maine's two-stage hierarchy of LDM analysis. Results for a new test data set, not used to build the model were 75%–100% accuracy (Davies et al. In press).

[7] The percentage of river and stream miles assigned to each ALU classification in Maine is: Class AA/A-49%; Class B-51%; Class C- 0.4%.

Maine's biological criteria models. After testing multiple statistical modeling techniques (e.g., k-means clustering, Two-Way Indicator Species Analysis, multivariate ordination), the use of best professional judgment of expert aquatic biologists and construction of a set of hierarchical linear discriminant models was selected as the most promising approach to accomplish both technical and regulatory policy goals.

Maine's tiered ALUs and calibration process for benthic macroinvertebrate samples utilizing professional judgment actually predated the formalization of the BCG, and development of the BCG was in fact based, in part, on Maine's approach to biological assessment and biological criteria (Davies and Jackson 2006). The calibration approach in Maine was similar to that described in section 4.1, except that professional judgment was used to place streams into Maine's designated ALU classes (Class A, Class B, Class C) instead of into BCG levels. Maine's tiered ALUs provide an ecologically descriptive gradient of condition tiers, with detailed definitions, to express the expected goal condition for each class. These clearly articulated goals provided the "guiding image" (Poikane et al. 2014; Willby 2011) for biologists to assign samples to classes. Maine DEP developed a set of multivariate linear discriminant models to predict the expert site assessment (Davies et al. 1995; Shelton and Blocksom 2004; State of Maine 2003; Davies et al. In press). The description of the model-building data set below is modified from Shelton and Blocksom (2004):

> The MEDEP [MDEP] originally developed the linear discriminant models based on 145 rock basket samples collected from across the state and representing a range of water quality during 1983–1989. They recalibrated the models in 1998 using a much larger macroinvertebrate database with a total of 376 sampling events (Davies et al. 1999). The final step involved assigning each of the 376 sites in the database to one of four *a priori* groups using the quantifiable measures.

MEDEP also conducts biological assessments of stream algal, wetland macroinvertebrate, and wetland phytoplankton and epiphytic algal assemblages (Danielson et al. 2011, 2012). MEDEP used Maine's narrative biological criteria and the BCG as the foundation of biological assessment models for stream algae, also using the LDM approach outlined here (Danielson et al. 2012). A first step in model-building was to empirically compute tolerance values for algal and macroinvertebrate species that had been collected in Maine's monitoring program. After computing tolerance values, the species were grouped into the BCG framework's sensitive, intermediate, and tolerant attribute groups. MEDEP then modified the model BCG framework for stream macroinvertebrates for stream algae and wetland macroinvertebrates, describing how those assemblages empirically respond to anthropogenic stressor gradients. MEDEP used those modified BCG frameworks and tolerance metrics along with the narrative biological criteria and other metrics to build predictive biological assessment models for the additional assemblages. MEDEP has completed LDM statistical models to predict ALU attainment for both stream algal and wetland macroinvertebrate community data. These models currently are used to help interpret narrative biological criteria. Following adequate testing and standard public review protocols, MEDEP will amend the Maine Biological Criteria Rule[8] to include the stream algal and wetland macroinvertebrate models as numeric biological criteria.

8 See Code of Maine Rules, MEDEP, Chapter 579, http://www.maine.gov/dep/water/rules/index.html. Accessed February 2016.

To define *a priori* groups for stream macroinvertebrates, biologists were given data from a set of sites and asked to place the sites into Maine's use classes based on the biological data only (Willby 2011). This set of sites was then used as the calibration data (or "learning" data) for an LDM. The objective of the discriminant model is to replicate ("predict") the professional judgment of the panel of biologists. The excerpt below describes how MEDEP biologists assigned calibration sites to Maine's three classes and to NA (from Davies et al. In press):

> Maine's statutory classes are goal-based and thus do not necessarily correspond to actual biological condition of streams in Maine so legislatively assigned classes could not be used to define groups ... As an alternative approach to defining stream classes, we used "expert knowledge/prior experience" to identify response signals (to different levels of human disturbance) for 30 quantifiable measures of macroinvertebrate community structure (Table 24 below). This classification process was then followed by validation using objective methods to confirm that the *a priori* groupings were, in fact, statistically distinguishable. This approach has been well developed (Anderson 1984; Press 1980). Discriminant analysis and function derivation does not have to rely on classes that only occur in nature. As long as classes are statistically distinct and their members possess a Gaussian distribution within a class, then most assumptions are met (Anderson 1984). To establish *a priori* groups, MDEP biologists, along with independent biologists from other states, and the private stakeholder sector, evaluated benthic macroinvertebrate community data for each stream sample (without knowing site locations or pollution influences) and assigned samples to an aquatic life condition category. The methodology was based on the degree to which each biologist found the sampled community conformed to one of the narrative aquatic life criteria (Class AA/A, B, C; or NA if the community assemblage did not conform to the narrative criteria of the lowest class) as described in the statute and accompanying definitions (Shelton and Blocksom 2004). The panel of biologists received limited habitat data (e.g., depth, water velocity, substrate composition, temperature) in order to evaluate the intrinsic biotic potential of the sampled habitat, but biologists had no knowledge of the site locations, or degree of human disturbance.

Biologist's Classification Criteria

Each biologist reviewed the sample data for the values of a list of measures of community structure and function. Criteria used by biologists to evaluate each measure are listed in Table 24. In 64% of the cases, there was unanimous agreement among the independent raters, and in an additional 34% of the samples, two of the raters were in agreement and one had assigned a different classification. In three of the rated samples, there was disagreement among all three raters (2%).

Table 24. Maine Biologists' Relative Findings Chart Using Macroinvertebrates (Source: Davies et al. In press)

Measure of Community Structure	Relative Findings by Water Body Class			
	A	**B**	**C**	**NA**
Total Abundance of Individuals	often low	often high	variable to high	variable: often very low or high
Abundance of Ephemeroptera	high	high	low	low to absent
Abundance of Plecoptera	highest	some present	low to absent	absent
Proportion of Ephemeroptera	highest	variable, depending on dominance by other groups	low	zero
Proportion of Hydropsychidae	intermediate	highest	variable	low to high
Proportion of Plecoptera	highest	variable	low	zero
Proportion of *Glossoma*	highest	low to intermediate	very low to absent	absent

Measure of Community Structure	Relative Findings by Water Body Class			
	A	B	C	NA
Proportion of *Brachycentrus*	highest	low to intermediate	very low to absent	absent
Proportion of Oligochaetes	low	low	low to moderate	highest
Proportion of Hirudinea	low	variable	variable	variable to highest
Proportion of Gastropoda	low	low	variable	variable to highest
Proportion of Chironomidae	lowest	variable, depending on the dominance of other groups	highest	variable
Proportion of *Conchapelopia* & *Thienemannimyia*	lowest	low to variable	variable	variable to highest
Proportion of *Tribelos*	low to absent	low to absent	low to variable	variable to highest
Proporation of *Chironomus*	low to absent	low to absent	low to variable	variable to highest
Genus Richness	variable	highest	variable	lowest
Ephemeroptera Richness	highest	high	low	very low to absent
Plecoptera Richness	highest	variable	low to absent	absent
EPT Richness	high	highest	variable	low
Proportion Ephemeroptera Richness	highest	high	low	zero
Proportion Plecoptera Richness	highest	high	low	low to zero
Proporation Diptera Richness	low to variable	variable	highest	variable to high
Proporation Ephemeroptera & Plecoptera Richness	highest	high	low to variable	low to absent
EPT Richness divided by Diptera Richness	high	highest	low to variable	lowest to zero
Proporation Non-EPT or Chronomid Richness	lowest	low	intermediate to high	highest
Percent Predators	low	low	high to variable	high to variable
Percent Collectors, Filterers, & Gatherers divided by Percent Predators & Shredders	high	highest	low	lowest
Number of Functional Feeding Groups Represented	variable	highest	variable	lowest
Shannon-Weiner Generic Diversity	low to intermediate	highest	variable to intermediate	lowest
Hilsenhoff Biotic Index	lowest	low	intermediate	highest

Once these groups were determined subjectively and independently by three biologists, univariate and multivariate analysis of variance (ANOVA and MANOVA, respectively) confirmed that the assigned groups were in fact statistically distinct. Following establishment and statistical validation of the groups, MEDEP applied additional analyses to evaluate the necessity to develop stratified models to account for natural factors, such as geographic location and stream size. The uni- and multivariate analyses (cluster analysis, multidimensional scaling, and principle components analysis, in part) suggested that a physically or geographically stratified model for Maine was not warranted. To determine variability in expert judgment assignments, a new test data set was assigned to *a priori* groups by two non-MEDEP biologists, yielding an average concurrence with MEDEP biologists' assignments of 80%. Furthermore, as a check against potential circularity in the model (i.e., "this site looks good, so this must be what good sites look like"), MEDEP chose 27 minimally disturbed sites based on non-biological criteria. These sites were not originally used in the expert assessment or to build the model. This reference data set was used to determine the success of the model to assign them to Class A conditions. These sites had no known point sources and land uses were characterized as 97% forested (3% logged); 2% crop; and 1% residential, industrial, or commercial.

Next, statistical methods and expert judgment were used to identify 26 biological community variables from a list of over 400 variables using stepwise discriminant analysis and iterative backward selection procedures to best assess attainment of the biological goals in the state's ALUs, and to best predict membership of an unknown stream sample to one of the four water quality classes (A, B, C, and NA). These were the methods used by Maine; for alternative approaches to variable selection and optimizing group separation, see Van Sickle et al. (2006). The 26 variables are in Table 25 (four original variables were discontinued following recalibration of the model). Linear discriminant functions were developed from the 26 quantitative macroinvertebrate variables. The discriminant functions determine the probability that a site belongs to a given water quality class. Using a linear optimization algorithm to calculate the discriminant function coefficients, multivariate space distance was minimized between sites within a class, while the distance between classes was maximized. Note that three variables used as predictors in the second-stage models were not calculated directly from the biological data, but instead were probabilities of group membership reported by the First Stage (four-way) discriminant model (see below).

The final, overall discriminant function is calculated using one four-way model and three two-way models. First, using only nine variables and calculated coefficients, the four-way model calculates the probability (range 0.0–1.0) that a site fits into each of the three attainment classes (AA/A, B, or C) and the non-attainment class (NA). The resultant probabilities are then transformed and used as variables in the three two-way models (Table 25). Use of the second stage, two-way models significantly improves the predictive accuracy of the overall model.

Table 25. Measures of community structure used in linear discriminant models for Maine (from MEDEP 2014; State of Maine 2003). Means refer to the mean of three rock baskets sampled at each site.

Model	No.	Measure
First Stage (four-way) model	1	Total mean abundance
	2	Generic richness
	3	Plecoptera mean abundance
	4	Ephemeroptera mean abundance
	5	Shannon-Wiener generic diversity (Shannon and Weaver 1963)
	6	Hilsenhoff Biotic Index (Hilsenhoff 1987a, 1987b)
	7	Relative Chironomidae abundance
	8	Relative Diptera richness (Diptera richness/generic richness)
	9	Hydropsyche mean abundance
Class C or Better model	10	Probability (A+B+C) from First Stage Model
	11	Cheumatopsyche mean abundance
	12	EPT:Diptera richness ratio
	13	Relative Oligochaeta abundance
Class B or Better model	14	Probability (A+B) from First Stage Model
	15	Perlidae mean abundance
	16	Tanypodinae mean abundance
	17	Chironomini mean abundance
	18	Relative Ephemeroptera abundance
	19	EPT generic richness
	21	Sum of mean abundances of: *Dicrotendipes*, *Microspectra*, *Parachironomus,* and *Helobdella*
Class A model	22	Probability of Class A from First Stage Model
	23	Relative Plecoptera richness (Plecoptera richness/generic richness)
	25	Sum of mean abundances of *Cheumatopshyche*, *Cricotopus*, *Tanytarsus,* and *Ablabesmyia*
	26	Sum of mean abundances of *Acroneuria* and *Stenonema*
	28	Ratio of EP generic richness (EP richness/14; 14 is maximum)
	30	Ratio of Class A indicator taxa (Class A taxa/7)

Note: Variable numbers are not sequential; variables 20, 24, 27, and 29 were discontinued following re-parameterization of the model.

The three two-way models further refine the discrimination among classes AA/A, B, or C. These models distinguish between a given class plus any higher classes as a group and any lower classes as a group (i.e., Classes AA/A + B + C vs. NA; Classes AA/A + B vs. Class C + NA; Class AA/A vs. Classes B + C + NA) as depicted in Figure 21, and model performance is shown in Table 26 below (MEDEP 2014; State of Maine 2003; Davies et al. In press). The two-way models are not strictly independent of the four-way model, because they use output probabilities of the four-way model as predictor variables.

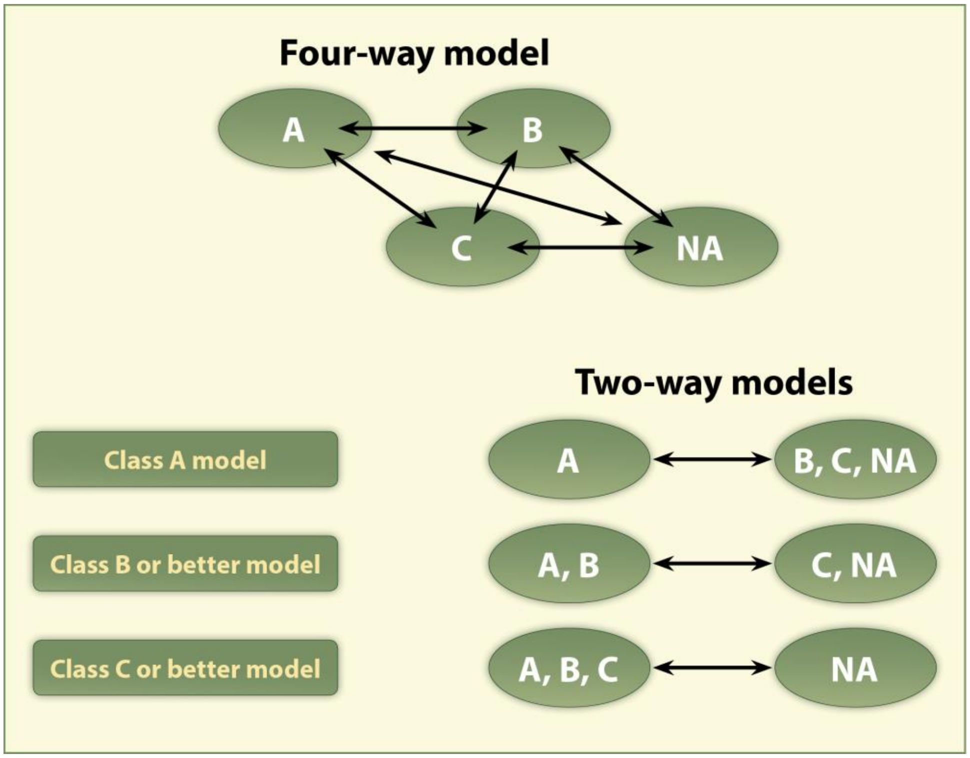

Figure 21. Schematic of four-way and two-way model relationships used by Maine DEP to refine the discrimination among classes (Source: MEDEP 2014).

Table 26. Classification of stream and river sites by two-way linear discriminant models for three classifications. Numerical entries represent the percent of sites classified from *a priori* classes (row) into predicted classes (columns). Therefore, diagonals are % correct classification.

Final A Classification		
Model Predicted Class		
***A priori* class**	**Class A**	**Classes B,C, or NA**
Class A	90.00% (108)	10.00% (12)
Classes B, C, NA	10.28% (26)	89.72% (227)
Final B or Better Classification		
Model Predicted Class		
***A priori* class**	**Class B or better**	**Classes C or NA**
Class B or better	96.57% (225)	3.43% (8)
Classes C, NA	11.43% (16)	88.57% (124)
Final C or Better Classification		
Model Predicted Class		
***A priori* class**	**Class C or better**	**NA**
Class C or better	96.07% (293)	3.93% (12)
NA	14.71% (10)	85.29% (58)

Note: Number in parentheses indicates the number of sites.

Once the probability that a site belongs to a certain class is calculated, the Maine Biocriteria Rule describes the assessment process the Department follows to conclude whether the site attains the minimum standards of its assigned classification (MEDEP 2014; State of Maine 2003). In order to determine whether a site attains at least Class C or is in non-attainment, the probability outcome using the "Class C or better model" is used. If the probability is greater than 60%, then the sample attains Class C or higher, but if it is less than 40% then the site is in non-attainment. If a site falls within 40%–60%, then best professional judgment is used to determine whether the site attains Class C, does not attain Class C, or is indeterminate of Class C. For any site found to be indeterminate, additional monitoring is scheduled in order to make a decision.

Those samples that attain Class C are then tested for Class B attainment using the probability of Class B outcome from the "Class B or better model." If the probability is greater than 60%, then the sites are deemed to attain at least Class B status. Those values below 40% are now considered to be sites that attain to Class C. If a value falls between 40% and 60%, then the outcome is indeterminate of Class B. If the site designated ALU is Class A or Class B, then additional monitoring is conducted to determine to which attainment class the site belongs.

When the probability outcome for a site is 60% or greater using the Class B or better model, it is then tested using the "Class A Model." If the probability of Class A is 60% or greater, then the site attains class A standards. If the value is 40% or less, then the site attains to Class B. If the value is between 40% and 60%, the finding is indeterminate of Class A (though it does attain Class B). Additional sampling will be required if the designated use of the site is Class A. Maine's WQS state that sites determined to attain the standards of the next higher class must be reviewed and considered for re-classification to the next higher class in order to maintain the higher water quality conditions that are being achieved (State of Maine 2004).

The LDM provides a probability of membership result. It explains model performance on a particular sample and can be used to assess the strength of the model decision. Additionally, each of variables can be examined to determine the strength of their contribution to the decision. After the LDM predicts the class attained by a site, a provision in MEDEP regulations (State of Maine 2003) allows for professional judgment to make an adjustment to the evaluation. Any adjustment may be made using analytical, biological, and habitat data. Professional judgment also may be employed when the condition of the stream does not allow for the accurate use of the linear discriminant models. Such factors may include habitat influences (e.g., lake outlets, impounded waters, substrate characteristics, tidal waters), sampling issues (e.g., disturbed samples, unusual taxa assemblages, human error in sampling), or analytical and sample processing issues (e.g., subsample vs. whole sample analysis or human error in processing) (MEDEP 2014; State of Maine 2003).

4.4 Automation of Decision Models

Any of the BCG decision models described above (sections 4.1–4.3) can be automated in databases, spreadsheets, or other commonly available software. Multimetric models have been incorporated into spreadsheet formulas and relational databases (e.g., Environmental Data Acquisition System [EDAS] and many state databases). Discriminant models and other statistical tools can also be coded in R and combined with a database or interactive web pages. More recently, several BCG multiple attribute decision models have been incorporated into MS-Access® applications.

For example, user-friendly automated models have been developed in Microsoft Excel® for the Upper Midwest (Gerritsen and Stamp 2012) and Northern Piedmont region of Maryland (Stamp et al. 2014). Additionally, the Little River Band of Ottawa Indians (LRBOI) has been using the Excel spreadsheets for the Upper Midwest BCG models to obtain BCG level assignments for all of their fish and macroinvertebrate samples from the lower Big Manistee watershed.

Geospatial database technology has advanced in recent years and shows promise for application in water quality management programs, including condition assessments. For example, Maine's discriminant model is incorporated into Maine's Oracle® relational database that is fully georeferenced and linked to the state's spatial database. The state's spatial database and selected, quality assured environmental data, including biological criteria assessment results, are publicly accessible via Google Earth.[9] Linkage between traditional databases that report biological assessment outcomes, and geo-spatial databases connected to natural bio-geophysical factors and disturbance parameters at multiple spatial scales, represent the growing edge of the emerging science of biological assessment.

4.5 Conclusion

A core objective of BCG calibration, from conceptualization to quantification, is to explicitly and transparently link science with management decisions in using biology to interpret ALU goals. This linkage can lead to enhanced stakeholder understanding and engagement in public decision making on goal setting and in assessing current conditions in relation to the ALU goals. However, information on stressors, their sources, and mechanism will be needed to identify actions to restore degraded waters and protect current conditions. Chapter 5 provides a conceptual framework, or template, to assist states in identifying the primary stressors and their sources and mechanisms of action, that impact their waters. This framework can be used by the states to organize data and information on watershed characteristics, hydrologic modifications, and stressors related to ALU goals.

[9] http://www.maine.gov/dep/gis/datamaps/index.html#blwq. Accessed February 2016.

Chapter 5. The Generalized Stress Axis

The x-axis of the BCG, the GSA, conceptually describes the full range, or gradient, of anthropogenic stress that may adversely affect aquatic biota in a particular geographic area. It is a theoretical construct that in application has been defined by states using known, quantitative stress gradients typically representing a portion of the stressors impacting a water body. The GSA provides a template for development of a quantified stress axis using available databases. Since the BCG curve represents the *in-situ* response of the resident biota to the sum of the stressors to which they are exposed, the GSA should be developed for the same geographic area and water body type for which the BCG is to be developed.

Once quantified, a GSA can serve several purposes. First and foremost, it can be used in development of decision rules for BCG model calibration. Second, the GSA and its underlying data can be used to inform management decisions and assess outcomes. Key applications of a GSA include:

Guide to selection of samples to be used in BCG decision rule development:

- Guide the selection of sites from a data set to ensure that the assessed sites cover as wide and full a range of stressors as possible, within the limits of the data set (see Chapter 3, section 3.3.1).
- Guide the assignment of different taxa to the different tolerance categories specified in the BCG (see Chapter 3, section 3.3.2).

Better link management decisions and outcomes:

- The data collected for developing a stress gradient might be used to help identify and rank sources and stressors within a region, watershed (e.g., 8- or 12-digit hydrologic unit code (HUC8 or HUC12, respectively)), and/or catchment[10] and improve the linkage between biological goals and management actions. Ideally, an improved connection between biological condition and stressors will assist state agencies in prioritizing sources and stressors for action, select effective BMPs, and track improvements. This application will likely occur after BCG development and require causal analysis (e.g., CADDIS; Suter et al. 2002; Norton et al. 2015).
- The data collected in development of the GSA might also be repurposed to inform additional management tools. For example, field-based stressor-response relationships can be used to help develop benchmarks for ALU (protective thresholds for contaminants or excess nutrients or conductivity; e.g., Cormier and Suter 2013; Cormier et al. 2013; USEPA 2011a). In addition, data analyses that describe the distribution of stressors that occur naturally can be repurposed to define background conditions.

This chapter describes the conceptual foundation of the GSA; discusses technical issues to be considered in developing a GSA for specific geographic areas and water body types; and, provides an overview of some approaches for quantifying a GSA.

[10] *Catchment* is defined as an incremental watershed that drains directly into a stream reach and excludes upstream areas. See: http://nhd.usgs.gov/. Accessed February 2016.

To date, GSAs have been used to develop decision rules to assign sites to BCG levels using known stress gradients and available regional, state, and/or county data (as described in first two bullets above). Some of these GSA applications were explained in the case studies in Chapter 3; they include quantitative gradients based on use of land cover indicators as surrogates for stressors (Minnesota, Alabama; see section 3.3.1.1), and an ordinal gradient based on the sum of cumulative stressors present at a site (Connecticut; see section 3.3.1.2). However, a systematic review and testing of the full suite of potential technical approaches to define and apply a GSA to BCG development has not been conducted. Opportunities in the future may include piloting methods for application of national, regional, or watershed scale data and methods to support state efforts to define and quantify the GSA. Examples of sources of data include EPA's National Aquatic Resource Surveys,[11] the StreamCat data set[12] (Hill et al. 2015), and EPA Office of Research and Development's watershed integrity indicators and map of the ecological condition of watersheds across the country (Flotemersch et al. 2015). Examples of methods that are currently available include the Healthy Watershed Methodology,[13] the Recovery Potential Screening tool (Norton et al. 2009),[14] the Analytical Tools Interface for Landscape Assessments (ATtiLA),[15] and the National Land Cover Database (NLCD).[16] Sources for both data and methods include the Watershed Index Online (WSIO)[17] and EnviroAtlas.[18]

5.1 The Conceptual Foundation of the Generalized Stress Axis

The purpose of this section is to provide a broad conceptual framework and terminology that describes the effects of human activities on biological communities and forms the basis for constructing a GSA. This framework can also be used to facilitate application of research to advance the development and application of the GSA as part of a quantitative BCG model.

The intent of the GSA is to reflect the cumulative degree of anthropogenic stress experienced by aquatic biota. Five major ecological factors that reflect environmental processes and materials determine the biological condition of freshwater aquatic resources: flow regime, water quality, energy source, physical habitat structure, and biotic interactions (Figure 22) (Karr and Dudley 1981). The first four of these factors (flow regime, water quality, energy source, and physical habitat structure) form the construct for a GSA. Appendix A-1 provides an organizing framework for a GSA and illustrates how a GSA might classify sites as high, medium, or no/low levels of stress for two general regions of the U.S., humid temperate and arid, based on these major factors.

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[11] http://www.epa.gov/national-aquatic-resource-surveys. Accessed February 2016.

[12] http://www.epa.gov/national-aquatic-resource-surveys/streamcat. Accessed February 2016.

[13] http://www.epa.gov/hwp. Accessed February 2016.

[14] http://www.epa.gov/rps. Accessed February 2016.

[15] http://www2.epa.gov/eco-research/analytical-tools-interface-landscape-assessments-attila-landscape-metrics. Accessed February 2016.

[16] http://landcover.usgs.gov/. Accessed February 2016.

[17] http://www.epa.gov/watershed-index-online. Accessed February 2016.

[18] http://www.epa.gov/enviroatlas. Accessed February 2016.

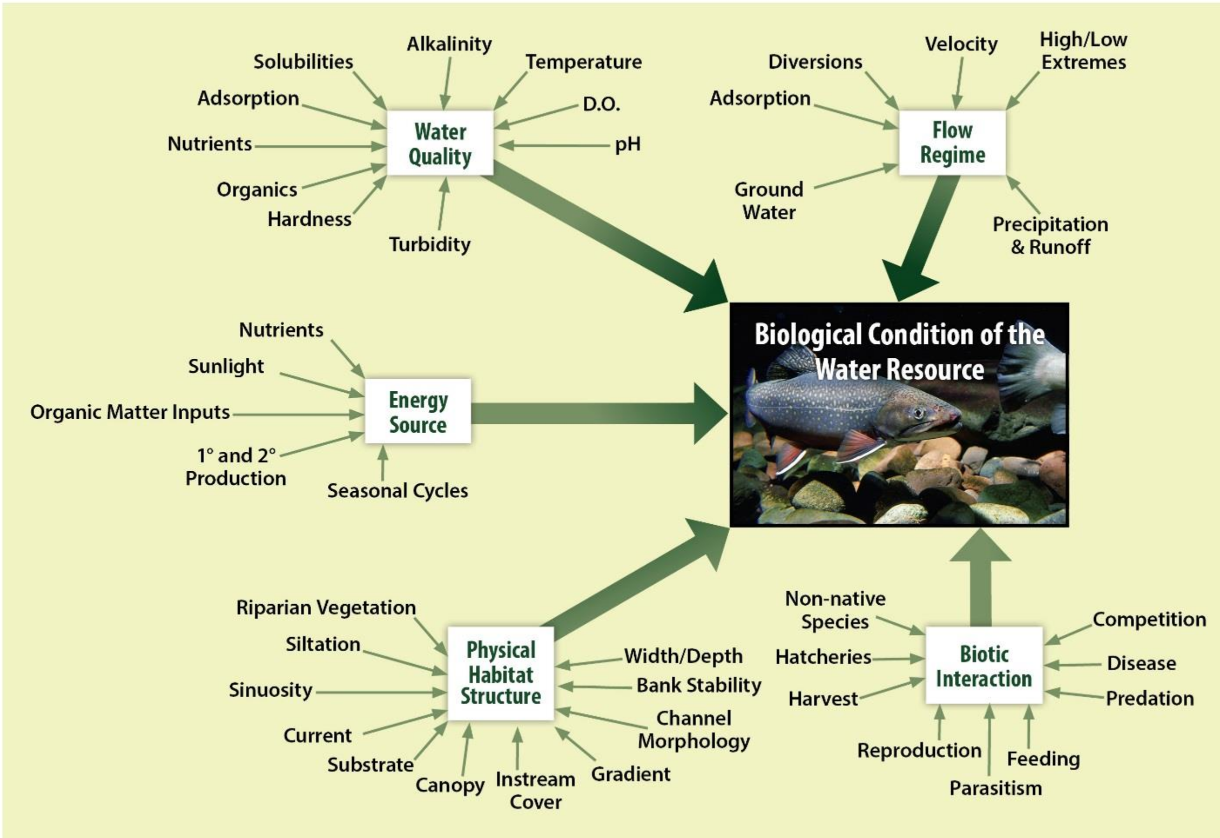

Figure 22. The five major factors that determine the biological condition of aquatic resources (modified from Karr and Dudley 1981). Four of the five factors, flow regime, water quality, energy source, and physical habitat structure, are the basis for the conceptual GSA as described in this document. The fifth factor, biotic interaction, is incorporated as part of the BCG y-axis levels and attributes.

An event or activity that alters one or more of these five factors is called a *disturbance*. Disturbances can occur outside of the stream and riparian zone (e.g., land use changes within the watershed, climate) or within it (e.g., dams, point source discharges). Ecosystems normally have some level of disturbances that occurs within a range of natural variability (e.g., Berger and Hodge 1998; White and Pickett 1985). Anthropogenic activities can cause disturbances that exceed the range of natural variability, and they are said to exert *pressure*[19] upon an aquatic system, or *state*, by altering ecosystem processes and materials, ultimately generating *stressors* that adversely impact biological condition (Niemi and McDonald 2004). The term *pressure* conceptually and mechanistically links larger scale landscape and hydrological alterations to the in-stream stressors that affect aquatic biota (Crain and Bertness 2006; Rapport and Friend 1979; Samhouri et al. 2010; Villamagna et al. 2013). Though different terminology is employed, the *Stressor-Exposure-System Response* paradigm (e.g., Barnthouse and Brown 1994) typically employed in water quality criteria development is comparable in that both conceptual models ultimately help accomplish the same objective—linking human activities to stressors to changes in biological condition (Figure 23) so action can be taken to protect or restore aquatic resources.

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[19] The use of the word pressure in this context has a well-established history in the European environmental literature. Pressure is a term originally proposed by the Organisation for Economic Co-operation and Development (OECD 1998) and used by the European Union in its Water Framework Directive (European Environment Agency 1999).

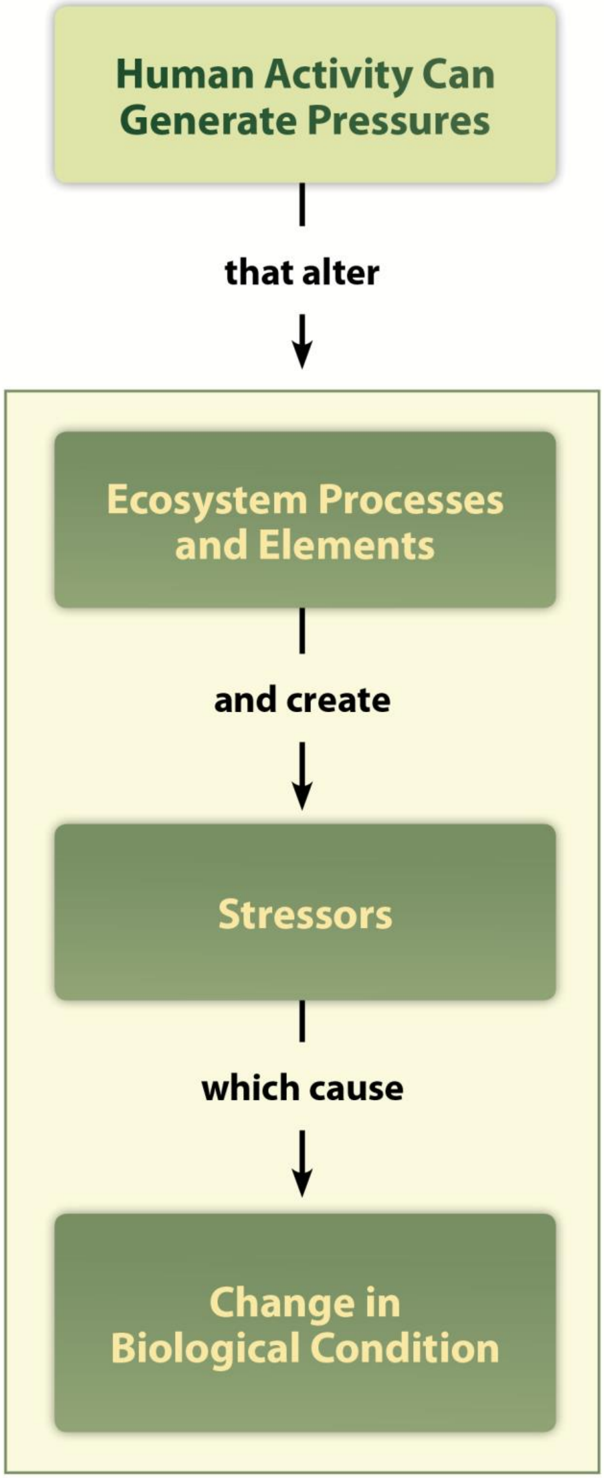


Figure 23. Human activities can cause *disturbances* in the environment that exceed the range of natural variability, generating *pressure* upon an aquatic system that results in altered *environmental processes and materials,* which, in turn, create *stressors* that adversely impact biological condition.

Stressors are the proximate causes of biological effects. They are the link between human activities and the change in biological condition (Figure 23). Stressors can co-occur in time and space when they are generated by the same human activity or source and/or any overlapping activity or source. Stressors may affect more than one aspect of biological condition, and a particular change in biological condition can also be the result of multiple stressors acting simultaneously. Since multiple stressors are usually present, the x-axis is intended to reflect their cumulative spatial/temporal co-occurrence in a GSA, much as the y-axis generalizes biological condition.

Point source discharges of pollutants were the dominant pressures to fresh waters addressed in the initial implementation of the CWA. While this pressure still exists today, water quality managers also face additional challenges stemming from in-stream hydrological modifications, forest harvest, agriculture, and urbanization, as well as emerging pressures associated with the inadvertent or deliberate introduction of invasive species (Ricciardi and MacIsaac 2000), the consequences of greenhouse gas emissions (e.g., Bierwagen et al. 2012), use of pharmaceutical products (Rosi-Marshall and Royer 2012; Rosi-Marshall et al. 2013), and even recreation (Bryce et al. 1999; Poff et al. 2002; Richter et al. 1997). Additionally, stressors can exert both direct effects on the biota and indirect effects through modification of habitat and interactions with other stressors (Karr and Dudley 1981; Karr et al. 1986; Poff et al. 1997; Slivitzky 2001) (Figure 24).

For example, a GSA that considers flow regime changes would consider many stressors and their interactions. Stream flows directly influence stream biota, but they also interact in multiple ways with other in-stream factors including water quality parameters, such as DO and temperature. Altered stream flows are strongly associated with many habitat variables such as channel structure, erosion, bank instability, and lower base flows (Poff et al. 1997; Richter et al. 2003; Poff et al. 2010). All of these factors associated with the flow regime have the capability of affecting species distributions, abundances, life history traits, and competitive interactions (Greenberg et al. 1996; Kennen et al. 2008; Poff and Allan 1995; Poff et al. 1997; Robson et al. 2011; Walters and Post 2011).

Many of the changes to the natural flow regime can be attributed to human activities, such as dam creation, channelization, and impervious surfaces, along with associated removal of natural vegetation, water extraction, and loss of surface water storage capacity (e.g., wetlands) (Poff et al. 1997). Altered flow regimes are also the result of changing climate, with changes observed in precipitation and runoff amounts, seasonal patterns, and timing, frequency, and intensity of large storms (Frich et al. 2002; Karl and Trenberth 2003; Poff et al. 2002). Still, flows vary naturally, and it can be difficult to distinguish anthropogenic disturbance from the range of variation produced by natural processes (e.g., see review by Berger and Hodge 1998). All of these issues should be considered when developing a GSA that reflects the stress associated with flow regime changes.

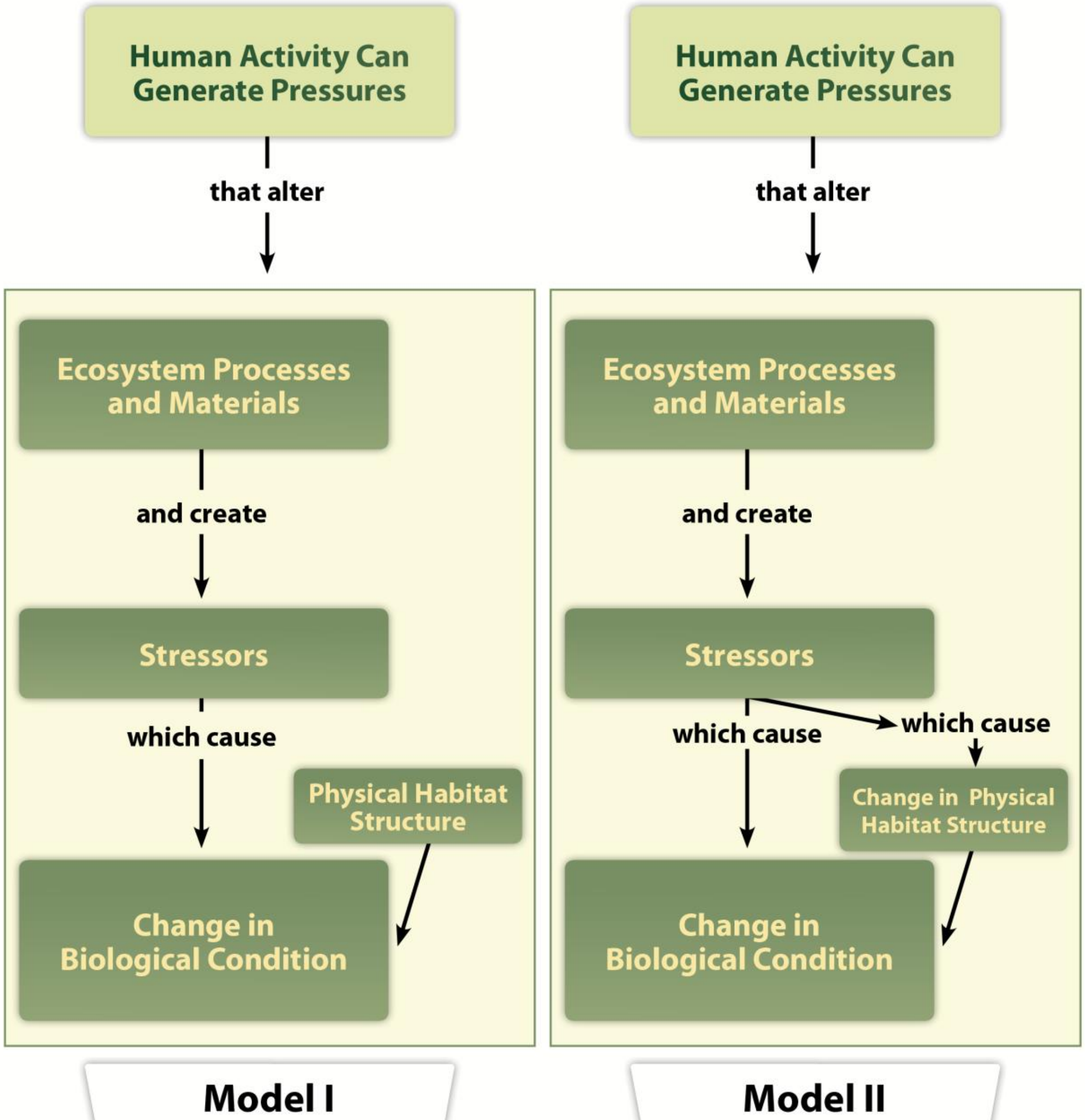


Figure 24. Hierarchical effects of disturbance. When assessing the relationship between stressors and biological effects, one of two implicit models is assumed. Model 1—the biota at a site are determined by the environmental covariates characteristic of the habitat. The stressors associated with a human-related disturbance directly influence biota. Model 2—the biota at a site are determined by the environmental characteristics of the site. However, the stressors associated with a human-related disturbance influence both the physical habitat structure and the biota itself. Consequently, the biological effects reflect the combined direct effects of the stress and the disturbance-mediated habitat alteration (From: Ciborowski et al. unpublished). Comprehensive and integrated monitoring data (biological, chemical, physical) coupled with causal assessment will help distinguish direct from indirect effects (USEPA 2013a).

5.1.1 Technical Issues in Developing a Generalized Stress Axis

This section discusses some of the technical issues to be considered in defining a GSA, including temporal and spatial scales, multiple stressors, legacy effects, and predicted impacts of climate change on aquatic systems. The concepts of spatial and temporal scale are critical issues in adequately defining the GSA. Pressures, stressors, and their effects on biota (e.g., biotic response) operate at different spatial and temporal scales (Glasby and Underwood 1996). Stressors are expressed over temporal and spatial scales ranging from a one-time, localized event (pulse event; Bender et al. 1984) to long-term chronic exposures occurring continuously (press events) over vast landscapes. Additionally, stressors may be introduced through diffuse or point sources delivered from upstream in the channel or watershed, or laterally from riparian, floodplain, or upland sources. Pollutants can also be delivered to a stream, river, lake or wetland from above through atmospheric sources, or below from groundwater sources. Activities in the watershed or along the water body corridor will influence the connectivity and integrity of the water resource. Additionally, climate change can exacerbate the intensity of local stressors (e.g., more heavy rainfalls can produce increased runoff and sediment load).

As discussed previously, human activities can produce multiple stressors, which in turn will affect biological condition. Stressors can interact with one another to create a synergistic response, behaving in an additive or multiplicative manner; they also may counteract one another. The steady accumulation of small pressures in watersheds results in cumulative effects, which add to the challenges of characterizing, evaluating, and managing stressors.

The influence of individual stressors on biological condition in specific water bodies can be particularly difficult to disentangle because each stressor potentially exerts indirect and direct forces. The complexity of interactions among stressors makes it difficult to identify single stressor-single biological effect relationships (Hodge 1997; Noss 1990; Vander Laan et al. 2013). Stressor identification is one causal assessment approach useful for identifying the stressors that cause biological effects (USEPA 2000; Norton et al. 2015).[20]

However, when sufficient data are available, quantitative modeling approaches can be used to describe the complex relationships between pressures, stressors, and their effects on the biota. Niemeijer and deGroot (2008a, 2008b) advocated summarizing the interactions among stressors to create causal networks as a means of better understanding the complex relationships between pressures and their ultimate effect on the biota, and this approach has been applied to streams with qualified success. Allan et al. (2012) used Bayesian Belief Network analysis to characterize the effects of sedimentation on macroinvertebrates in agricultural streams in the U.S. Midwest and in New Zealand affected by sedimentation due to grazing and forestry practices. Riseng et al. (2010, 2011) used Structural Equation Modeling to document relationships between stress and stream biota. They determined that land use effects in total were more important influences on metrics of fish and invertebrate biota than effects of point source discharges.

The concept that human activities produce multiple stressors provides the foundation for one common approach to describing an overall gradient of stress using land cover information as a surrogate for stressor information. In this approach, the GSA is developed using broadly defined, relatively easily measured factors that produce many stressors simultaneously (e.g., amount of urban development or road density in a catchment). Mapping the distribution of pressures, for example land uses associated

[20] See also http://www3.epa.gov/caddis/. Accessed February 2016.

with particular human activities, has proven to be an effective way of documenting the location of possible sources that produce the stressors that lead to biological degradation (Allan et al. 2013; Brooks et al. 2009; Danz et al. 2005, 2007).

Stressor indicators can be developed from such measures as population density, proportion of land devoted to agriculture or urban development, total miles of roadway, or quantities of water used/released (e.g., Allan et al. 2013; Host et al. 2005, 2011; Hunsaker et al. 1992; Jones et al. 1999, 2001; O'Neill et al. 1988, 1997; Riitters et al. 1995, 1996, 1997). The advent of improved remote sensing, digital technology, and the ability to map land uses has provided an important tool for documenting the location and extent of pressures on the landscape. This approach has been used effectively to assess watershed and coastal conditions such as in the Laurentian Great Lakes for decades where Danz et al. (2005, 2007) and Allan et al. (2013) documented the distribution of the composite stress contributed by human activity throughout the Great Lakes (Figure 25). A simplified form of the Danz et al. (2005) system, the Watershed Stress Index (Host et al. 2011), is currently used to report on the condition of Great Lakes watershed, including tracking progress towards achieving the overall purpose of the binational Great Lakes Water Quality Agreement "to restore and maintain the physical, chemical and biological integrity of the Great Lakes Basin Ecosystem."[21] Allan et al. (2013) used expert assessment to delineate threats to the biological integrity of the Great Lakes themselves. Host et al. (2011) mapped the distribution of watersheds in which specific groups of biota were at least and at greatest risk of degradation due to urban and agricultural pressures.

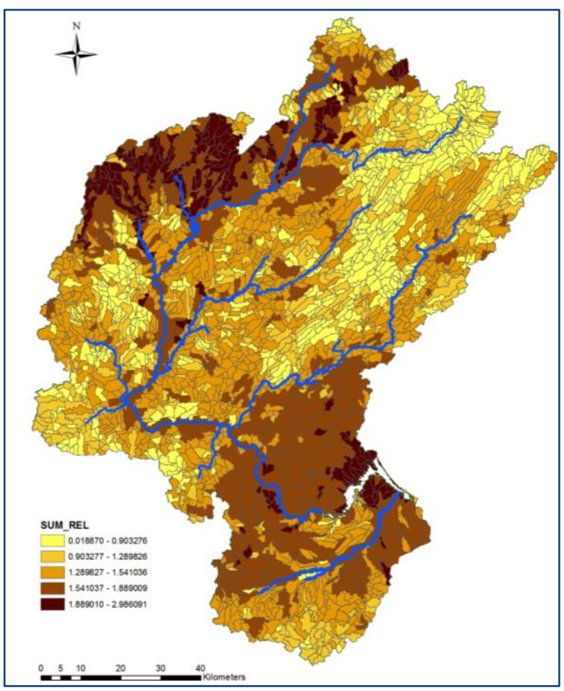


Figure 25. Cumulative stress within the St. Louis River watershed, a tributary to Lake Superior. Darker shading indicates increased stress. The stress score is based on the cumulative sum of % agricultural land use, population density, road density, and point source density. Values were each normalized to a 0–1 scale before summation. This index was used to calibrate water quality responses to stress in the St. Louis River Area of Concern (Bartsch et al. 2015). (Map by Tom Hollenhorst, EPA, Mid-Continent Ecology Division)

[21] https://www.ec.gc.ca/grandslacs-greatlakes/default.asp?lang=En&n=70FFEFDF-1. Accessed February 2016.

However, although land use can be a useful general pressure indicator, practices within a given land use category can change over time, which may reduce or increase the stressors that are produced by that land use. Local variables can exert important influences on biological conditions that are not captured by remote sensing or other land cover data alone. For example, the incidence of tile drainage is generally not mapped; drainage intensity has increased in some areas of the Midwest resulting in increased annual flows in ditches related to reduced evaporation off of land surfaces (Blann et al. 2009). Miltner (2015) used extensive biological, stressor, and pressure (agricultural practices) data in Ohio and demonstrated that conservation measures have contributed to improved environmental conditions in Ohio headwater streams. Miltner (2015) concluded "that stream physical habitat clearly influences water quality, and therefore structural measures that improve habitat function in channelized streams and drainage ditches are a necessary component of efforts to combat eutrophication." Analyses such as these would not be possible without the accumulation of substantial monitoring data collected at a higher spatial resolution (Blann et al. 2009; Miltner 2015). Additionally, documenting biological conditions at the local reach and watershed scale makes it apparent that broad scale use of indicators such as land cover are not in themselves adequate predictors of biological impairment in specific water bodies. The scale of application is a critical factor—important stressors that act at the local reach and watershed scale can be missed.

An additional caveat in using land cover as a sole basis for GSA development is that the indicators are typically based on current land uses although some types of past land use patterns are available as mapped information. Many human activities in watersheds leave permanent or semi-permanent changes, termed "legacy effects." For example, persistent contaminants such as DDT, PCBs, PAHs,[22] and metals can end up in sediments, and they may be resuspended or buried permanently, depending on the depositional environment. Excess phosphorus may be buried in lake or pond sediments. In eastern U.S. Piedmont and Appalachian highlands, stream valley morphology has changed permanently in many places due to historic land use changes from the colonial period to the present: from initial clearing, to colonial and early American hydropower development, early agriculture, subsequent agricultural abandonment and forest regrowth, followed by recent suburban development (e.g., Maizel et al. 1998; Walter and Merritts 2008). These legacies may account for intermittent stressors in the form of contaminants, nutrients, and sediments that can be eroded and resuspended from historic sedimentation during storm events, or permanent stressors in the form of hydrological modifications or sedimentation. Documenting previous land use and expanding monitoring programs to include appropriate parameters will assist in detection of these stressors.

Regardless of the information used in defining a GSA, the impact of climate change will increasingly need to be taken into account. Climate change is a widespread disturbance that is capable of moving the system outside its natural range of variation, even in the absence of other anthropogenic disturbances, by elevating air and water temperatures, altering flow regimes through changes in the seasonality of precipitation, altering soil moisture regimes, and through changes in the frequency and intensity of storm events and fires (IPCC 2014; Melillo et al. 2014). The effects of changing climatic conditions, whether considered naturally or anthropogenically driven, are superimposed on other anthropogenic stressors generally leading to an exacerbated effect (c.f. Comte et al. 2013; Palmer et al. 2009; Hoegh-Guldberg et al. 2007; Arnell 1999). In general, water quality is likely to be negatively impacted by effects of climate change through altered flow regimes leading to higher peak flows and lower base flows. Altered flow regimes in turn influence extremes in water temperature, DO concentrations, changes in

[22] DDT: dichlorodiphenyltrichloroethane; PCB: polychlorinated biphenyl; PAH: polycyclic aromatic hydrocarbon

biogeochemical processing, and biotic assemblage structure and function that these factors regulate (Melillo et al. 2014). The effects of heavy downpours are exacerbated by impervious surfaces, leading to greater sediment, contaminant, and nutrient loading. Appendix A-2 provides examples of stressors and potential indicators of climate change under low, medium, and high stress scenarios for humid and arid regions. The BCG with well-defined biological indicators (y-axis) and stress indicators (x-axis) can be used to determine current baseline conditions and track changes in parameters that are associated with climate change, such as flow and temperature.

5.2 Development of a Generalized Stress Axis

In preparation for BCG development (see Chapter 3, sections 3.2 and 3.3), the process to develop a GSA for a specific geographic area and water body type includes a series of steps: classifying sites to reduce natural variability; identifying undisturbed or minimally disturbed conditions; and identifying indicators and the data that will be used to define the gradient of stress.

The first step in GSA development is to classify the aquatic resource (e.g., biogeographic regions, basins, biological considerations) (Herlihy et al. 2008; McCormick et al. 2000; Van Sickle and Hughes 2000; Waite et al. 2000). Classification is also an important component of biological assessment program development (see section 3.2.1.1). The purpose of classification is to reduce variability in natural conditions that can contribute to or influence stressors and biological assemblages. Features such as latitude, climate, geology, and landforms can explain the dominant patterns of variation in stressors across large regions (e.g., Herlihy et al. 2008). These broad-scale classification systems can be supplemented by local-scale features (e.g., slope, groundwater seeps) that can contribute to site-scale patterns in biotic assemblages (Hawkins and Vinson 2000; Pyne et al. 2007; Snelder et al. 2004, 2008; Van Sickle and Hughes 2000).[23]

A second step in GSA development is characterizing undisturbed or minimally disturbed conditions for a particular area. This characterization is the benchmark against which areas to be evaluated will be compared (as discussed in section 3.2.1.1), allowing for development and calibration of indices such as the mIBI and O/E assessment models. For most state biological assessment programs for streams, this step involves use of the state's reference site database. An important consideration when selecting reference sites is whether the reference sites represent undisturbed, minimally disturbed, or least disturbed conditions (Hawkins et al. 2010; Herlihy et al. 2008; Hughes 1985, 1994; Hughes et al. 1986; Moss et al. 1987; Stoddard et al. 2008). In BCG development, descriptions of undisturbed and minimally disturbed reference conditions (e.g., BCG levels 1 and 2) are critical components of model calibration. In some places, calibration may be based solely on historic records or other sources of information. Like level 1 of the BCG, the "low stress" end of the stress axis is anchored in the "as naturally occurs" or undisturbed or minimally disturbed, condition (i.e., no/minimal anthropogenic stressors).

The third step is to identify indicators and data sets that will be used to define the GSA. The major environmental factors shown in Figure 22 can be used as prompts to identify indicators (e.g., Appendix A-3). When evaluating data sets to develop a GSA, it is important to bear in mind that the biological conditions will reflect effects of unknown sources and unmeasured stressors, as well as incorrectly

[23] A comprehensive review of recent classification systems is beyond the scope of this document. There is still much to be learned about how biotic effects from local vs. catchment scale disturbances differ between catchments that are largely disturbed, and those that are relatively undisturbed (see review by Johnson and Host 2010).

characterized data sets. In this regard, the GSA is only as robust as the data upon which it is based. Characterizing to the extent possible the degree of uncertainty around the stressor-response (i.e., effect) relationships is important. There will always be some level of unexplained variation. But, where relationships between stress, or stressors, and biological response are poorly predicted, further assessments should be conducted. For example, as mentioned above, legacy contaminants from long-defunct industrial activities are typically invisible to remote imaging, yet may wash out periodically in storm events. A water quality assessment conducted for screening purposes is unlikely to capture such rare events. Intensive, directed sampling is more likely to detect the contamination, possibly after determination that a downstream location is biologically impaired from unknown causes and historical land use records are researched.

As explained earlier, this document does not comprehensively review or evaluate the approaches available to define a GSA. The examples discussed below represent several approaches that have been used to define stress gradients and are intended to prompt ideas and enhancements.

5.2.1 Using Land Cover Measures as Stressor Indicators

One approach to quantify a GSA relies upon land cover data. The land cover indicators serve as surrogate indicators for stressors, typically multiple stressors associated with a specific land use. Many human activities that cause stress in aquatic systems can be summarized in land cover delineations. Because land cover can be expressed as a fraction or percent of a watershed, catchment, or zone within the catchment (e.g., riparian corridors), using land cover data provide an obvious initial approach for summing land uses for an overall index of pressure. Land cover data generally do not include information on legacy sources and stressors unless intentionally mapped, nor do the data usually incorporate in-stream measures of water quality or habitat quality. Thus, the methods that rely solely on land cover should be regarded as the “first cut” tool in a toolbox that may contain multiple approaches. If stress-response relationships are poorly predicted by land cover data, subsequent analyses should include a more complete portfolio of stressors that contain both local habitat and water quality variables, as well as potential legacy pressures. Although remote sensing is a useful coarse focus, stressors and their effects on the biota can vary substantially.

The simplest land cover-based GSA is comprised of one, or the sum of several, land covers calculated for the catchment of each aquatic sampling point in the database being used. For example, in the Maryland Piedmont, percent impervious surface was used as a single stressor gradient because of the extent of urban and suburban land use throughout the mid-Atlantic Piedmont (see Chapter 3, sections 3.3.1.1 and 3.3.2.1). As another example, developers of a BCG for fish assemblages in Minnesota lakes used a GSA composed of a simple sum of percentages of urban, agricultural, and mining lands (section 3.3.2.1).

The above land cover-based GSAs do not differentially weight various land uses (as measured by land cover) in terms of their effects on aquatic biota. For example, impervious surface strongly affects stream hydrology, habitat quality, and biology (e.g., Stranko et al. 2008) and effects of agricultural land use depend on its intensity and local agricultural practices. An alternative method, the landscape development intensity index (LDI), weighs the intensity of multiple land uses in a study area (Brown and Vivas 2005). The LDI is a measure of human activity based on a development intensity measure derived from non-renewable energy use in the surrounding landscape. The LDI is calculated using all nonrenewable forms of energy (e.g., electricity, fuels, fertilizers, pesticides, and water (both public water supply and irrigation) (Brown and Vivas 2005)) used directly or implicitly in various land use classifications. Land uses are classified, and an intensity factor is assigned to each land use type (Table 27).

Table 27. Land use classification and intensity factor (LDI coefficient) for Florida landscapes (modified from Brown and Vivas 2005)

Land Classification	Intensity Factor (LDI coefficient)
Natural system	1.00
Natural open water	1.00
Pine plantation	1.58
Recreational/open space – low intensity	1.83
Woodland pasture (with livestock)	2.02
Improved pasture (without livestock)	2.77
Improved pasture – low intensity (with livestock)	3.41
Citrus	3.68
Improved pasture – high intensity (with livestock)	3.74
Row crops	4.54
Single-family residential – low density	6.9
Recreational/open space – high intensity	6.92
Agriculture – high intensity	7.00
Single-family residential – medium density	7.47
Single-family residential – high density	7.55
Mobile home (medium density)	7.70
Highway (2-lane)	7.81
Low intensity commercial	8.00
Institutional	8.07
Highway (4-lane)	8.28
Mobile home (high density)	8.29
Industrial	8.32
Multi-family residential (low-rise)	8.66
High-intensity commercial	9.18
Multi-family residential (high-rise)	9.19
Central business district (average 2-stories)	9.42
Central business district (average 4-stories)	10.00

The LDI has been used as a human disturbance gradient for wetlands (Brown and Vivas 2005; Chen and Lin 2011; Lane 2003; Mack 2006, 2007; Reiss 2004, 2006; Reiss and Brown 2005, 2007; Surdick 2005; Vivas 2007; Vivas and Brown 2006), streams (Brooks et al. 2009; Fore 2003, 2004; Harrington 2014; Stanfield and Kilgour 2012), and lakes (Fore 2005). It has also been used for coral reefs (Oliver et al. 2011). Figure 26 shows application of the LDI for coral reefs. Land use indices similar to the LDI were used to develop BCG calibrations for Minnesota and Alabama (see section 3.3.1.1).

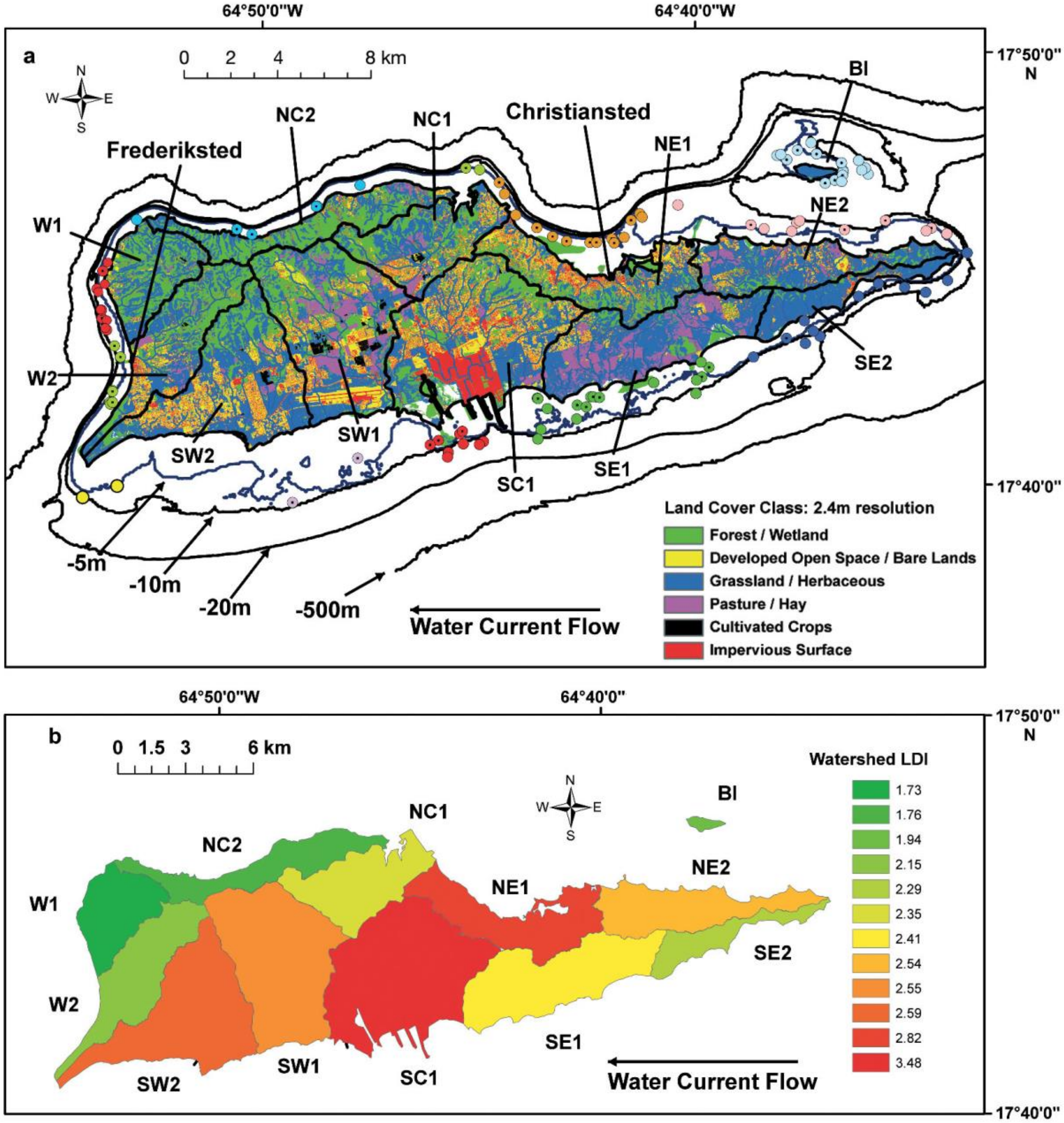


Figure 26. LDI applied to St. Croix watersheds and associated coral stations (Source: Oliver et al. 2011). Top figure shows land use/land cover and EPA coral reef stations. Land use/land cover used in the analysis is shown at 2.4 m resolution. Bottom figure show the watershed LDI values on a green– yellow–red continuum, where green indicates the lowest human disturbance and red indicates the highest. Watershed abbreviations: BI: Buck Island; NC: North Central; NE: Northeast; SC: South Central; SE: Southeast; SW: Southwest; W: West.

Nationally, the LDI has been mapped at HUC12 watershed as part of the WSIO data library using publicly available data from 2001. The WSIO contains mean, median, standard deviations, and sum of values for empower density (derivation of LDI) for a HUC12 watershed, its riparian zone, and hydrologic connected zone. Currently the WSIO data set is being updated nationally with the most recent NLCD data and should be available for use in near future.

5.2.2 Ranking Sites by Summing Stressor Indicators

Another approach to develop a GSA is to tally the number of stressor indicators observed at a particular site and establish a method to score the results. Many examples of this approach have been used across different regions, spatial extents, and ecosystem types (Chow-Fraser 2006; Uzarski et al. 2005). This approach entails identifying observed human activities and observed stressors (and their sources if information is available) and summing them to produce an overall index that can then be used to place sites in order from least to most stress.

The first step for the ordinal approach involves identifying and quantifying, for each site in a biological monitoring database, the relevant data available, including data on sources, in-stream measured water quality, riparian condition, land cover, riverscape alterations, known point source discharges, and observed nonpoint sources. For instream measures, it is important to distinguish non-detects (known and effectively absent) from not sampled (unknown; no data). A conceptual diagram of sources, stressors, mechanisms, and effects is helpful in organizing the information (e.g., Norton et al. 2015).

In the simplest implementation, each stressor indicator is evaluated as being present (1) or absent (0) at a site. The results are added to produce a score for each site. In the Connecticut case example (section 3.3.1.2), stressor indicators included reduced natural land cover, developed land, impervious surface, total chloride (a measure of total point source discharge), and four metals (copper, iron, nickel, zinc). Scoring in the case example was not simply 0–1; some stressor scores could range on an ordinal scale of 0–3, depending on the concentration or intensity of a given stressor. The results were used to divide sites into five overall stress categories ranging from "least stressed" to "severe stressed." The resultant gradient helped identify potential most-stressed, least stressed, and intermediately stressed sites in the BCG development data set. It is important to reiterate that the stress information was hidden from the expert panel during its deliberations.

For development of the BCG in Minnesota, MPCA developed a disturbance index (the HDS) that combined scores associated with land use metrics with additional indicators. The index includes eight primary metrics, which include measures of watershed land use, stream alteration, riparian condition, and known permitted discharges. The disturbance index scores can range from 1, representing completely altered and heavily stressed streams, to 81, representing nearly pristine watersheds. The HDS is described by MPCA (2014e) (see section 3.3.1.1, Table 7). Alabama DEM developed a similar index (see section 3.3.1.1, Table 8).

5.2.3 Using Statistical Approaches to Combine Stressor Indicators

In the U.S. Great Lakes coastal region, principal components analysis (PCA) was used by a team of researchers and investigators participating in the Great Lakes Environmental Indicators (GLEI) Project[24] (Niemi et al. 2007) to reduce over 200 variables into a single gradient, applying measures of anthropogenic pressures as surrogate measures of stressors (Danz et al. 2005). The Danz approach individually considered six different indicators of pressure: agriculture, atmospheric deposition, land cover, human population, point sources, and shoreline alteration. The GLEI team used a watershed-based approach to reflect the premise that the environmental effects of these activities in coastal watersheds can influence environmental conditions in downstream coastal ecosystems. The first principal component from the analysis explained 73% of the variance in the agricultural-chemical (Ag-Chem) variables (reflecting land use, agricultural chemical use, and agricultural-influenced nutrient and

[24] http://glei.nrri.umn.edu/default/default.htm. Accessed February 2016.

sediment loading) and was interpreted as an overall gradient in stressors across the basin (Figure 27). Environmental effects such as changes in water quality, fish assemblage metrics, and bird abundances were strongly correlated with scores of this stressor gradient, providing verification that the statistically extracted PCA was biologically meaningful (see description of this project by Niemi et al. 2007). The GLEI researchers created a flow diagram (Figure 28) that details their steps for quantifying a stressor gradient (modified from Danz et al. 2005).

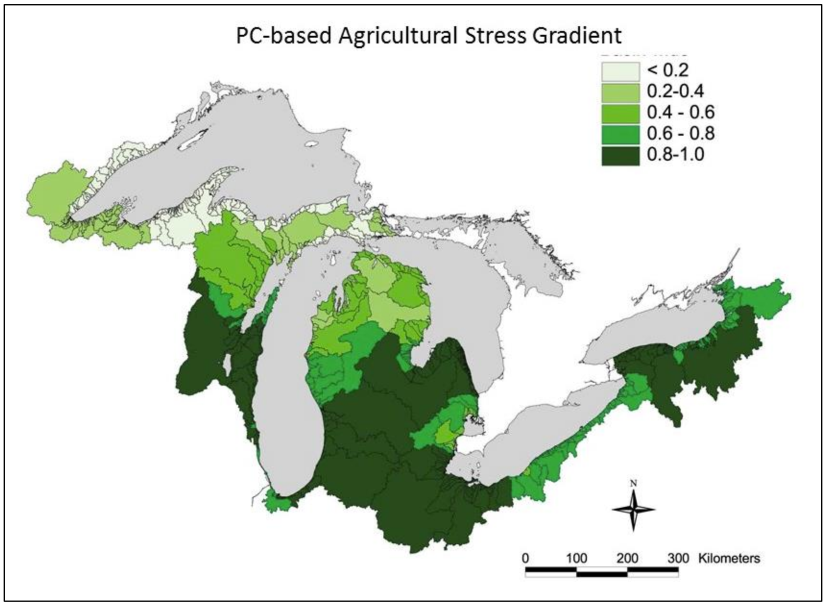


Figure 27. The first principal component of the agricultural variables for the U.S. Great Lakes basin. Darker shading indicates greater amounts of agriculture (Source: Danz et al. 2005).

While the pressure-stressor model eventually developed for the Great Lakes coastal region was visualized as a single gradient from low to high levels of stressors, different individual and combinations of stressors are expected to dominate in different regions. Furthermore, disaggregating the PCA into individual categories of stress could provide important information about potential mechanisms affecting the state of the system.

In addition to PCA, there are other statistical approaches to consider. For example, the use of non-metric multidimensional scaling (NMDS) provides a robust analysis. Unlike PCA, NMDS can deal with non-normal data, data of varying scales, and outliers in the data. Like PCA, NMDS is a multivariate statistical analysis that one can use to look at multiple stressors at the same time to create the GSA.

Biological data can also be used to statistically combine stressor indicators into a GSA. For example, Wang et al. (2008) used Canonical Correspondence Analysis (CCA) to derive the relationship among the biota and stressor and land use data and weight their disturbance index. They then plotted the calculated disturbance index against fish IBI scores and percentages of intolerant individuals, dividing the disturbance index values into five tiers. The first tier was the maximum disturbance index value at which the fish measures did not show an obvious decline. The remaining four tiers were determined by dividing the remainder of the disturbance index values into even categories. Use of biological data ensures that the stressor indicators will be biologically relevant. However, this approach can introduce some circularity into the analysis if the indicators of biological quality are the same as those used to develop the BCG.

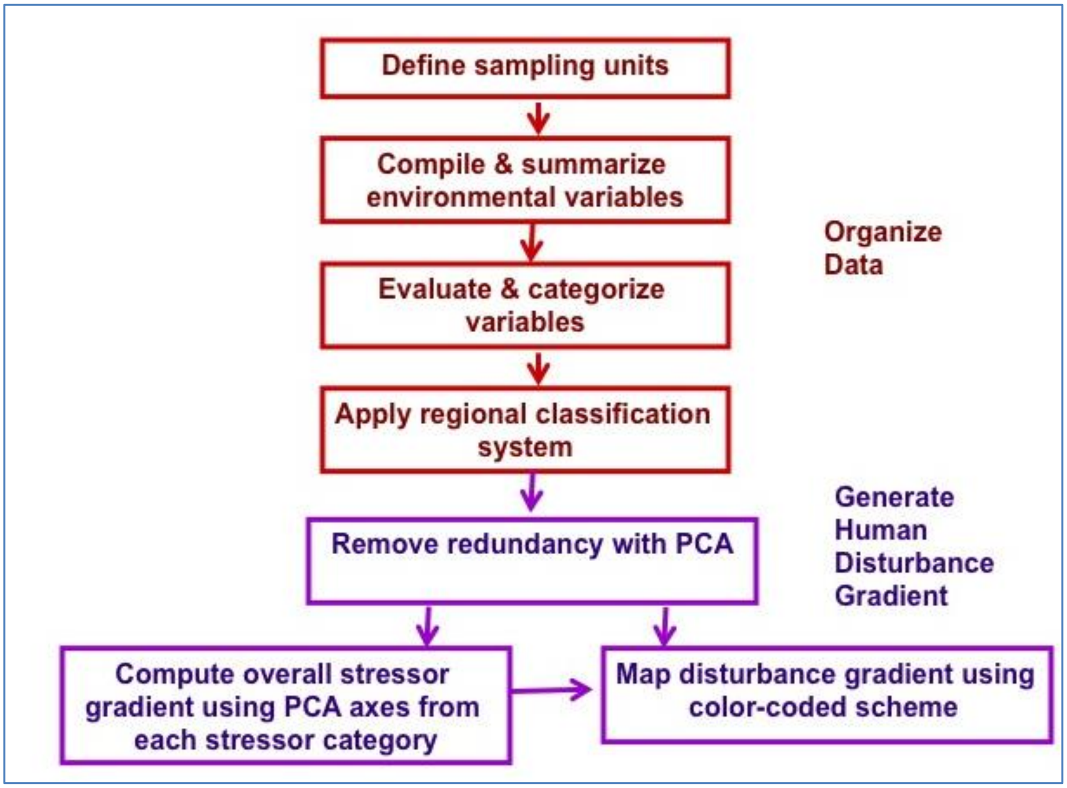


Figure 28. Flow diagram detailing the steps used by GLEI researchers in quantifying their stressor gradient (modified from Danz et al. 2005).

Stressor gradients like that developed by GLEI, or others as referenced above, can be developed at different spatial scales. The GLEI study assessed 5,971 watersheds comprising the Great Lakes basin. Watershed sizes (areas) were lognormally distributed, with a median watershed area of 4.3 km^2 and a mean watershed size of approximately 86.7 km^2 (Ciborowski et al. 2011). However, the gradient can be applied and scaled as needed to other geospatial units. For example, Nieber et al. (2013) conducted this same analysis for watersheds of the north shore of Lake Superior, and Bartsch et al. (2015) scaled their analysis to watersheds of the St. Louis River estuary to assess relationships between stressors and water chemistry.

5.3 Linking the Science with Management Actions

A quantitative BCG model provides a framework for assessing baseline biological condition and, with systematic monitoring, can be used to track changes in biological condition. Ideally, a well-defined GSA and the stressor effects and biotic response models underlying it can be used in conjunction with causal assessments to better link biologically-defined management goals to the actions taken to protect or restore the biological conditions.

A stressor can be traced back to its source or tracked forward to the biological effect via a causal pathway (Figure 29). For example, stream banks that become destabilized due to removal of riparian plants could be the source of excess fine sediment to a stream. Erosion by high flows is the mechanism by which the excess fine sediments are generated, and the resulting in-stream siltation is the stressor. Smothering of bottom substrate habitat and organism gills by these fine sediments are two mechanisms by which biota are exposed and adversely affected. Invertebrate mortality and fish emigration could be some of the environmental outcomes or changes in biotic condition. Further, degradation or loss of recreational fishing could be societal impact of these changes and may prompt a conservation or restoration effort depending upon the circumstances.

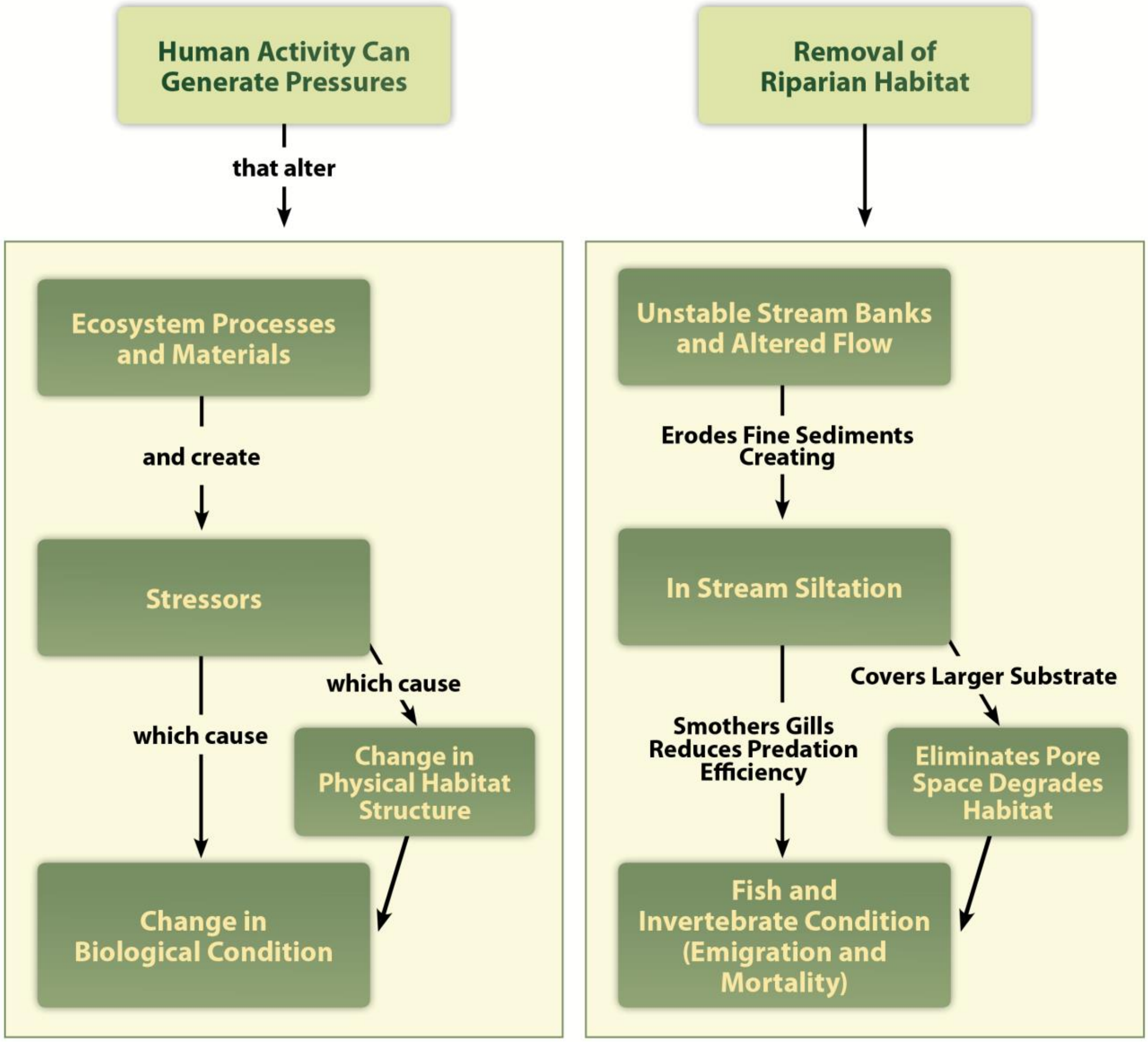


Figure 29. The specific stressor(s) and their intensity (the BCG x-axis—termed the GSA) are created by pressure(s) acting through specific mechanisms. BMPs can be implemented to prevent or reduce effect on the biota through restoration, remediation, and/or mitigation.

Actions can be taken that insulate the aquatic biota from the effects of anthropogenic pressures, helping to maintain or restore the ecological potential of an aquatic system. In the example above, re-establishing the riparian zone would stabilize the banks and prevent further erosion and unchecked flow into the stream. Appendix A-4 and MPCA (2015) provide examples of pressures linked to mechanisms and potential management actions that can mitigate the effect on biota.

Mechanistic processes operate between pressures and stressors and between stressors and their effects on biological condition (Figure 30; Appendices A-3 and A-4). Understanding these mechanisms and how they operate helps in predicting the potential effects of a particular management action. The BCG provides a framework for tracking and documenting incremental improvements in biological conditions resulting from implementation of a single BMP or combination of BMPs.

Integrating monitoring programs with frameworks like the BCG can improve understanding of how human activities, stressors, biological responses, and management actions are linked, providing feedback to guide management decisions. For example, Yoder et al. (2005) reviewed changes to fish assemblages over 25 years based on an intensive pollution survey designed to assess non-wadeable rivers in Ohio. They used the linkages between changes in point source pollution loadings, improvements in instream water quality, and reductions in the extent and severity of biological impairments to document the effectiveness of advanced wastewater treatment on a statewide scale beginning in the late 1970s. At that time the documented improvements in biological condition across all rivers and streams were almost solely in response to water quality-based NPDES permitting for point sources. Rivers predominantly impacted by nonpoint sources showed improvement over a longer timeframe where there was a concerted effort to apply BMPs over a wide enough region. Miltner (2015) was able to document widespread improvements in stream biota and water quality in smaller headwater streams in Ohio. Both of these studies were based on the state's routine biological monitoring and assessment of rivers and streams.

A well-defined GSA, and the underlying data set, can serve as a nexus between biological and causal assessments and provide a link between management goals and selection of management actions for protection or restoration. The basis of the BCG framework is that greater pressures can generate increased levels of stressors, and in turn, increased stressors are associated with reduced biological condition (Figure 30A and B). Typically, the stressors on aquatic systems increase as pressures increase, which results in a consequent decrease in biological condition. Effective management practices can target any point in the web of causal events, mitigating the effects of pressures and reducing stressors with resulting protection or improvement in biological condition (Figure 30C and D).

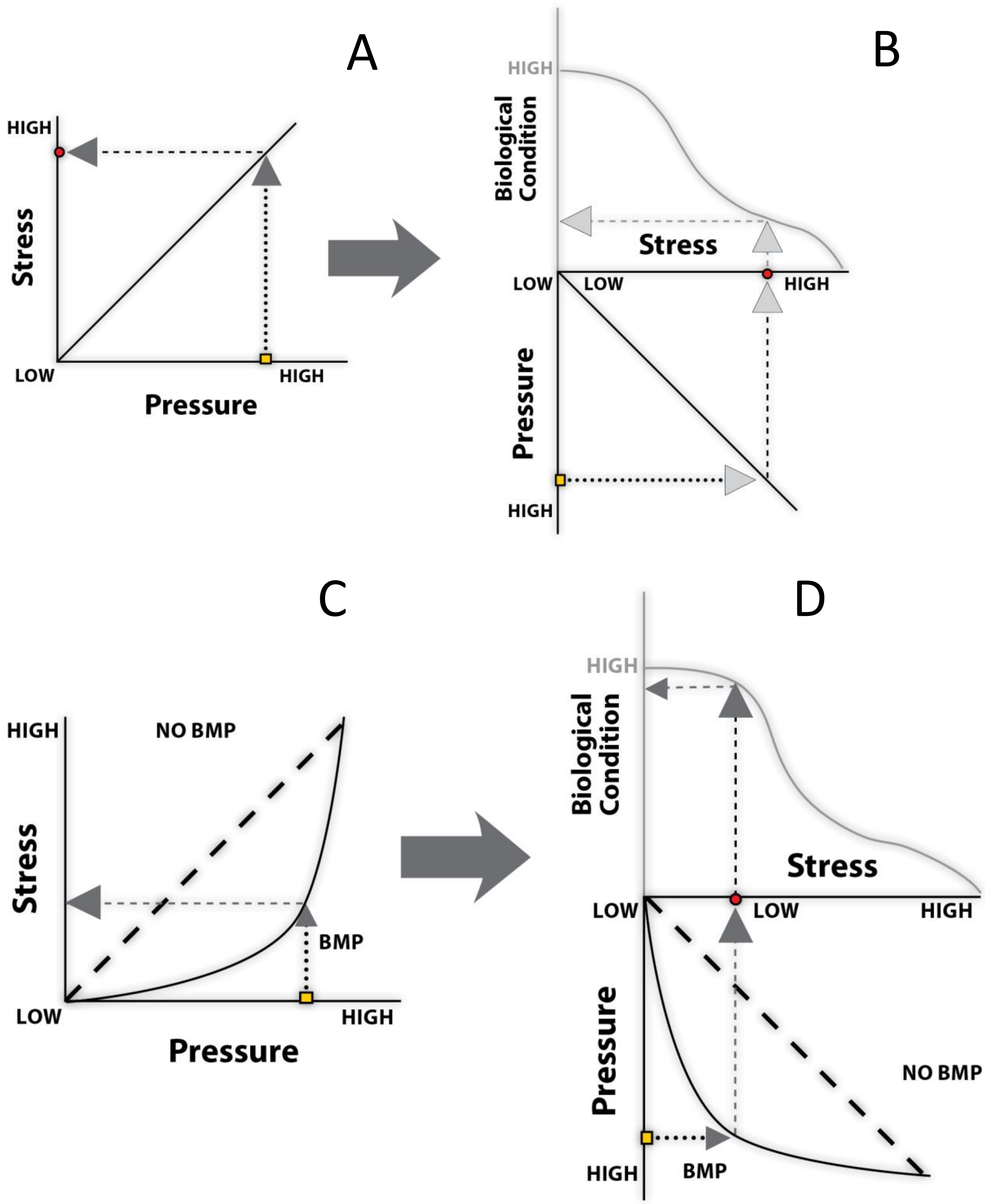


Figure 30. Conceptual Models (CM) A-B: Human activities can generate pressures, ultimately producing stressors (BCG x-axis) that adversely affect the aquatic biota (BCG y-axis). CM C-D: Implementation of a BMP can dampen the translation of pressures into the expression of stress and reduce the adverse effects on the biota.

5.4 Conclusions

Anthropogenic activities exert *pressures* on aquatic systems by altering ecosystem processes and materials and generating stressors that adversely impact biological condition. Many of these stressors co-occur in time and space, and effects on the biota are cumulative. The relationships between stressors and effects are complicated—stressors may affect more than one aspect of biological condition, and a particular change in biological condition can also be the result of multiple stressors acting simultaneously.

The conceptual GSA describes the range of anthropogenic stress experienced by aquatic biota in a particular geographic area. Once quantified, it is used in the development of the decision rules to assign sites to BCG levels (Chapter 3, section 3.3.1) and ensures that the BCG encompasses the full range of condition along a stress gradient. There is much complexity of interactions and effects from multiple stressors with varying effects on different biotic components of any aquatic system. The GSA represents the sum total of stressors and their sources in concept, but in implementation it is composed of multiple known, quantitative stress gradients that each represent a portion of the actual stress gradient to which the aquatic biota are exposed. The usefulness of the conceptual framework is to provide a template for as thorough and comprehensive a technical approach as possible to develop the BCG x-axis and relate level of stress to the BCG levels and attributes.

Additionally, developing a GSA that reflects the human activities and stressors in a particular geographical area helps in understanding how specific stressors are generated and how they affect biotic condition. The data generated in developing a GSA can be used to help identify and rank sources and their stresses in a particular area and inform management decisions on appropriate actions to protect or improve a water body. The case examples discussed in this chapter and in Chapter 3 illustrate how state and local governments have quantified a GSA as part of developing a BCG model for their specific region or watershed area. As further experience is gained and approaches to define and quantify the GSA evolve, EPA may supplement this document with additional information.

Chapter 6. Case Studies

The BCG can provide critical information to state water quality management programs at the watershed, statewide, and ecoregional scales. A comprehensive monitoring and assessment program is a critical aspect of implementation of the BCG to support water quality management programs. The same data and information that provide baseline condition assessments over time also can provide information to inform trend assessments and track incremental changes in condition. In conjunction with monitoring data, a BCG can be used to help address watershed-specific management needs such as detailed biological descriptions of designated ALUs, identification of high quality waters and impaired waters, and documentation of incremental improvements due to controls and BMPs. This information can also inform TMDL development. This chapter presents six case examples of how states, counties, or municipalities are using, or considering using, the BCG to support water quality management decision making.

The six case examples are:

- 6.1 Montgomery County, Maryland: Using the Biological Condition Gradient to Communicate with the Public and Inform Management Decisions
- 6.2 Pennsylvania: Using Complementary Methods to Assess Biological Condition of Streams
- 6.3 Alabama: Using the Biological Condition Gradient to Communicate with the Public and Inform Management Decisions
- 6.4 Minnesota: More Precisely Defining Aquatic Life Uses and Developing Biological Criteria
- 6.5 Maine: Development of Condition Classes and Biological Criteria to Support Water Quality Management Decision Making
- 6.6 Ohio: Tiered Aquatic Life Use Classes and Comprehensive Water Quality Management Program Support

6.1 Montgomery County, Maryland: Using the Biological Condition Gradient to Communicate with the Public and Inform Management Decisions

6.1.1 Key Message

Montgomery County helped to develop a BCG to better inform the public and county decision makers about a high quality watershed (e.g., undisturbed/minimally disturbed conditions) and the potential outcome of planned development. Local government decision makers were able to understand how these high quality streams compared to other streams in Montgomery County and Maryland. Development plans were modified to protect the streams and watershed and reduce environmental impacts, while allowing development to proceed.

6.1.2 Background: Early County Policy

In 1994, the Maryland-National Capital Park and Planning Commission (M-NCPPC) adopted the *Clarksburg Master Plan & Hyattstown Special Study Area*. The Plan established goals for development of Clarksburg, Maryland, at that time a mostly undeveloped area along a six to eight lane highway corridor outside the Washington, DC metropolitan area. The Plan's goals included development of the town with emphasis on maintaining farmland and open space and promotion of transit-oriented neighborhoods (M-NCPPC 1994). One critical objective of the plan was the protection of environmental resources while accommodating development, such as affording special protection to high quality stream systems, including tributaries to the streams and associated wetlands. The plan specified that development occur in four phases, with requirements that must be met in order for development to proceed from one phase to the next. This staging allowed for consideration of new data and information on the impacts of development on streams and rivers, as well as improvements in mitigation technology and changes in county, state, or federal policies or regulations that might affect implementation of the 1994 plan. For example, in 2008, the County revised the 1994 plan to meet the newly adopted state law requiring the use of Environmental Site Design (ESD) practices to minimize stormwater runoff throughout the county.


Figure 31. Ten Mile Creek, Maryland.

Development in one of the high quality areas slated for development, Ten Mile Creek (TMC) (Figure 31), was afforded special protection under the Master Plan. TMC, a subwatershed[25] of the Little Seneca Creek watershed, was assigned to stage four to ensure that the 1994 development plan could be reviewed and potentially adjusted based on relevant new data and information. This case example shows how the BCG was used to provide information on current conditions in TMC relative to other county subwatersheds and streams in *excellent*, *good*, *fair*,

[25] A subwatershed is the topographic perimeter of a stream catchment.

or *poor* condition. Information from the BCG was used in conjunction with other data to help inform the County Council in its deliberation on whether or not to adjust the stage four development plan.

6.1.2.1 Ten Mile Creek Subwatershed, Stream, and Tributaries

The TMC subwatershed, stream, and tributaries comprise a headwater stream system in which the majority of tributaries are small and spring fed. Abundant springs and seeps supply cold and clean water that supports a diverse community of fish, benthic macroinvertebrates, and amphibians (Boucher, personal communication, 2014) (Figure 32). The area is highly forested with a low level of impervious surface, < 1% to 3%. TMC is one of three reference watersheds remaining in the county and has supported *good* to *excellent* conditions based on a long term county data set using IBIs for benthic macroinvertebrates and for fish that were developed by the county (MCDEP 2012). TMC and its tributaries are adjacent to both Little Bennett Creek, a natural resource conservation management area, and to the county's agricultural reserve. The location of TMC provides not only a bridge between these two protected areas, but also a cost efficient opportunity to maintain natural flows, clean water, and high biological diversity, and provide for recreational use and appreciation by the public (Figure 33).

Figure 32. Important aquatic species in Maryland's Piedmont headwater streams. Salamanders (Long-tailed, Northern Dusky, and Northern Red); fish (Potomac Sculpin, Rosyside Dace, American Eel); insects (Sweltsa, Paraleptophlebia, Ephemerella).

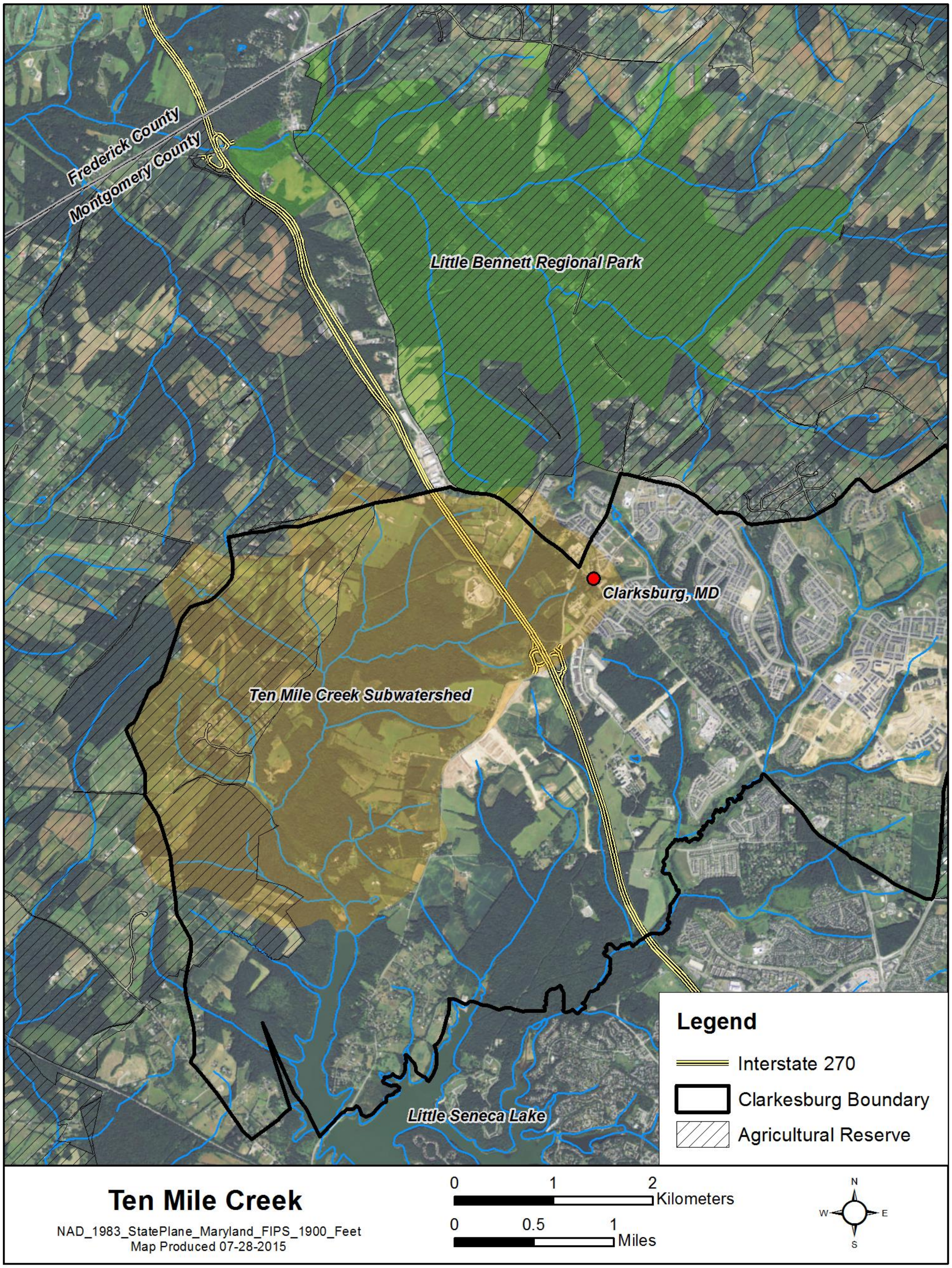


Figure 33. Clarksburg Area and Ten Mile Creek Subwatershed.

6.1.2.2 Monitoring the Impacts of Development

Beginning in 1994, the Montgomery County Department of Environmental Protection (MCDEP) monitored conditions throughout the Clarksburg development area as construction progressed. Analysis included evaluating the effectiveness of BMPs and regulations to minimize both the immediate impacts from construction and the longer term impacts from the subsequent development. Annual monitoring reports were published beginning in 2001 (e.g., MCDEP 2009, 2012). Initial monitoring found stream conditions in the Clarksburg development area ranged from *good* to *excellent* in most sensitive, high quality areas such as the TMC subwatershed. However, by the mid-2000s, the water quality at several good quality streams in the urbanizing areas began to degrade from *good* to *fair* (MCDEP 2009, M-NCPPC 2014a). In October 2012, the Montgomery County Council directed the County Planning Board to undertake a limited amendment of the 1994 Clarksburg Master Plan. Monitoring of earlier Clarksburg developments showed uncertainty about the ability to protect the sensitive environmental resources found in the stage four development area, such as TMC subwatershed, if full development were to occur according to the original 1994 plan.

A number of scientific analyses informed the development of the *Ten Mile Creek Area Limited Amendment to the Clarksburg Master Plan and Hyattstown Special Study Area*. County staff sought to use their extensive monitoring data to further characterize the watershed and to identify analytical ways to present information on the environmental status of County waters. Specifically, staff wanted to assess the current conditions in those waters and the expected changes that would occur in relation to further development in the area. In an effort to further characterize and assess incremental changes in local biological conditions, in 2013 the County embarked on the process of developing a BCG model for the Piedmont region of Maryland using both county and state data for fish and benthic macroinvertebrate assemblages (USEPA 2013b). Observations on the presence of salamanders were also incorporated where data were available. The presence of stream salamander species such as the northern dusky salamander, long tailed salamander, northern two-lined salamander, and the northern red salamander aided in confirming the high quality of streams.

6.1.3 Development of the Biological Condition Gradient

The County saw the BCG as one way to provide more detailed information on streams and their response to land use change. In 2013, scientists from agencies within the state, Delaware, Pennsylvania, Virginia, EPA, consulting groups, and academia convened as an expert panel to develop a BCG for the Northern Piedmont. The goal of this effort was to use data collected primarily from Montgomery County to develop a BCG model to describe changes in the biota in response to increasing stress in the landscape. For example, a BCG level 2 stream would be minimally disturbed and include the presence of native top predator fish (e.g., brook trout) as well as mayflies, stoneflies, and caddisflies. A BCG level 3 or 4 stream would include incrementally higher loss of sensitive species and an increased abundance of tolerant species (e.g., blacknose dace and northern two-lined salamander). A BCG level 5–6 stream would show an abundance of highly tolerant species (e.g., brown bullhead, tubificid and naidid worms).

Experts at the workshop were able to distinguish five distinct levels of biological condition for the Piedmont region within Montgomery County (BCG levels 2–6). There were no BCG level 1 sites. Most TMC sites ranged from a level 3+ to a level 4, although several sites (e.g., primarily headwater streams) were judged as very good quality (a level 2 rating). Narrative and numeric decision rules to consistently describe and quantify site assessments were developed based on mathematical set theory using the fuzzy logic method (Table 28, Table 29, Table 30) and taxa response relationships derived from the county data sets (Figure 34).

Table 28. Description of fish, salamander, and macroinvertebrate assemblages in each assessed BCG level. Definitions are modified after Davies and Jackson (2006).

BCG level 1	**Definition:** Natural or native condition—*native structural, functional and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability*
	Narrative from expert panel: There are no BCG level 1 sites within the Piedmont. All sites have some degree of disturbance, including legacy effects from agriculture and forestry from 100 to 200 years ago. Conceptually, BCG level 1 sites would have strictly native taxa for all assemblages evaluated (fish, salamander, benthic macroinvertebrates), some endemic species, and evidence of connectivity in the form of migratory fish.
	Fish: Examples of endemic species that might be present (depending on the size of the stream) include: Bridle Shiner, Brook Trout, Chesapeake Logperch, Maryland Darter, Trout Perch
	Macroinvertebrates: Sensitive-rare, coldwater indicator taxa such as the mayfly Epeorus, and stoneflies Sweltsa and Talloperla are expected to be present
BCG level 2	**Definition:** Minimal changes in structure of the biotic community and minimal changes in ecosystem function—*virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability*
	Narrative from expert panel: Overall taxa richness and density is as naturally occurs (watershed size is a consideration). These sites have excellent water quality and support habitat critical for native taxa. They have many highly sensitive taxa and relatively high richness and abundance of intermediate sensitive-ubiquitous taxa. Many of these taxa are characterized by having limited dispersal capabilities or are habitat specialists. If tolerant taxa are present, they occur in low numbers. There is connectivity between the mainstem, associated wetlands and headwater streams.
	Fish: Highly sensitive (attribute II) and intermediate sensitive (attribute III) taxa such as yellow perch, northern hog sucker, margined mad tom, fallfish and fantail darter are present, as are native top predators (e.g., brook trout). Migratory fish and amphibians (e.g., eel, lamprey, salamanders) are present or known to access the site. Long-tailed and Dusky salamanders are also good indicators, given a complimentary fish community. Non-native taxa such as brown trout or rainbow trout, are absent or, if they occur, their presence does not displace native trout or alter structure and function.
	Macroinvertebrates: Highly sensitive taxa are present—especially coldwater indicator mayflies, stoneflies, and caddisflies (e.g., Epeorus, Paraleptophlebia, Sweltsa, Tallaperla, and Wormaldia)—and occur in higher abundances than in BCG level 3 samples.

BCG level 3	**Definition:** Evident changes in structure of the biotic community and minimal changes in ecosystem function—*Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but intermediate sensitive taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system*
	Narrative from expert panel: Generally considered to be in good condition. Similar to BCG level 2 assemblage except the proportion of total richness represented by rare, specialist and vulnerable taxa is reduced. Intermediate sensitive-ubiquitous taxa have relatively high richness and abundance. Taxa with intermediate tolerance may increase but generally comprise less than half total richness and abundance. Tolerant taxa are somewhat more common but still have low abundance. Taxa with slightly broader temperature or sediment tolerance may be favored.
	Fish: Intermediate sensitive (attribute III) taxa such as fallfish and fantail darter are common or abundant. Taxa of intermediate tolerance (attribute IV) such as channel catfish, least brook lamprey, pumpkinseed and tessellated darter are present in greater numbers than in BCG level 2 samples. Some tolerant (attribute V) taxa such as mummichog and white suckers may be present, but highly tolerant taxa are absent. Pioneering species such as blacknose dace, creek chubs and white suckers may be naturally common in smaller streams. Migratory species such as American Eel may be absent. Two-lined salamanders may occur.
	Macroinvertebrates: Similar to BCG level 2 assemblage except sensitive taxa (e.g., Sweltsa, Tallaperla and Wormaldia) occur in lower numbers. Level 3 indicator taxa include the caddisfly Diplectrona, the mayfly Ephemerella and the stonefly Amphinemura.
BCG level 4	**Definition:** Moderate changes in structure of the biotic community and minimal changes in ecosystem function—*Moderate changes in structure due to replacement of some intermediate sensitive taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes*
	Narrative from expert panel: Sensitive species and individuals are still present but in reduced numbers (e.g., approximately 10%–30% of the community rather than 50% found in level 3 streams). The persistence of some sensitive species indicates that the original ecosystem function is still maintained albeit at a reduced level. Densities and richness of intermediate tolerance taxa have increased compared to BCG level 3 samples.
	Fish: 2 or 3 sensitive taxa may be present but occur in very low numbers (e.g., Blue Ridge Sculpin, Fantail Darter, Potomac Sculpin, Fallfish, Rosy-side Dace, River Chub). Taxa of intermediate tolerance (attribute IV) such as tesselated darter, least brook lamprey, longnose dace are common, as well as tolerant taxa like yellow bullhead, red-breast sunfish and bluntnose minnow. Level 4 streams may harbor two to three salamander species (Dusky, Red, and Two-lined).
	Macroinvertebrates: Sensitive taxa (including EPT taxa) are present but occur in low numbers. Taxa such as Diplectrona and Dolophilodes may occur, but other key taxa such as Ephemerella and Neophylax are absent. Taxa of intermediate tolerance (e.g., Baetis, Stenonema, Caenis, Chimarra, Cheumatopsyche, Hydropsyche) occur in greater numbers. Tolerant taxa such as Chironomini and Orthocladiinae are present but do not exhibit excessive dominance.

BCG level 5	**Definition:** Major changes in structure of the biotic community and moderate changes in ecosystem function—*Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials*
	Narrative from expert panel: Overall abundance of all taxa reduced. Sensitive species may be present but their functional role is negligible within the system. Those sensitive taxa remaining are highly ubiquitous within the region and have very good dispersal capabilities. The most abundant organisms are typically tolerant or have intermediate tolerance, and there may be relatively high diversity within the tolerant organisms. Most representatives are opportunistic or pollution tolerant species.
	Fish: Facultative species reduced or absent. Tolerant taxa like yellow bullhead, red-breast sunfish, and bluntnose minnow are common. Blacknose dace, creek chubs and white suckers may dominate. Two-lined salamanders might be the only salamander present.
	Macroinvertebrates: Highly sensitive macroinvertebrate taxa are usually absent and Chironomid midges (mostly tolerant Orthocladiinae and Chironomini) often comprised > 50% of the community in level 5 streams.
BCG level 6	**Definition:** Major changes in structure of the biotic community and moderate changes in ecosystem function—*Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials*
	Narrative from expert panel: Heavily degraded from urbanization and/or industrialization. Can range from having no aquatic life at all or harbor a severely depauperate community composed entirely of highly tolerant or tolerant invasive species adapted to hypoxia, extreme sedimentation and temperatures, or other toxic chemical conditions.
	Fish: Fish are low in abundance or absent, represented mainly by blacknose dace, green sunfish, bluntnose minnow, or creek chub.
	Macroinvertebrates: May be dominated by tolerant non-insects (Physid snails; Planariidae; Oligochaeta; Hirudinea; etc.)

Table 29. BCG quantitative decision rules for macroinvertebrate assemblages. The numbers in parentheses represent the lower and upper bounds of the fuzzy sets.

BCG Level 2	rule	
# Total taxa	> 17 (13–22)	
% Attribute II taxa	≥ 8% (5–10)	
% Attribute II+III taxa	≥ 50% (45–55)	
% Attribute II individuals	≥ 3% (2–5)	
% Attribute II+III individuals	≥ 60% (55–65)	
% Attribute V individuals	≤ 15% (10–20)	
BCG Level 3	**alt 1**	**alt 2**
# Total taxa	> 17 (13–22)	
% Attribute II+III individuals	≥ 40% (35–45)	
# Attribute II taxa	—	≥ 1 (0–2)
% Attribute II+III taxa	≥ 25% (20–30)	≥ 45% (40–50)
% Attribute V individuals	≤ 40% (35–45)	≤ 50% (45–55)
% Most dominant Attribute V individual	≤ 20% (15–25)	—
BCG Level 4	**rule**	
# Total taxa	≥ 15 (10–20)	
% Attribute II+III taxa	≥ 20% (15–25)	
% Attribute II+III individuals	≥ 10% (5–15)	
% Attribute V individuals	≤ 70% (65–75)	
% Most dominant Attribute V individual	≤ 60% (55–65)	
BCG Level 5	**rule**	
# Total taxa	≥ 8 (6–10)	
% Attribute V individuals	≤ 85% (80–90)	
% Most dominant Attribute V individual	≤ 70% (65–75)	

Table 30. BCG quantitative decision rules for fish assemblages in small (0.5–1.4 mi^2), medium (1.5–7.9 mi^2) and larger streams (> 8 mi^2). The numbers in parentheses represent the lower and upper bounds of the fuzzy sets. The mid-water cyprinid taxa metric is comprised of notropis, luxilus, clinostomus, and cyprinella, minus swallowtail shiners.

BCG Level 2	Small		Medium		Large
	rule	alt rule	rule	alt rule	rule
# Attribute I taxa	> 0 (present)		> 0 (present)		–
# Attribute I+II taxa	–		≥ 2 (1–4)		≥ 4 (2–6)
# Attribute I+II+III taxa	> 1 (0–3)	–	–		–
# Sensitive salamander taxa (if surveyed)	–	> 0	–	> 0	–
% Attribute I+II+III taxa	≥ 35% (30–40)		≥ 35% (30–40)		≥ 35% (30–40)
% Attribute I+II+III individuals	–		≥ 50% (45–55)		≥ 50% (45–55)
# Attribute VIt taxa	≤ 2 (1–3)		≤ 2 (1–3)		≤ 2 (1–3)
% Attribute VIt individuals	≤ 5% (3–7)		≤ 5% (3–7)		≤ 5% (3–7)
# Attribute X taxa	–		> 0		> 0
BCG Level 3	**Small**		**Medium**		**Large**
# Attribute I+II taxa	–		–		≥ 1 (0–2)
# Attribute I+II+III taxa	≥ 2 (0–4)		–		–
% Attribute I+II+III taxa	–		≥ 25% (20–30)		≥ 25% (20–30)
% Attribute I+II+III individuals	≥ 25% (20–30)		≥ 25% (20–30)		≥ 25% (20–30)
% Attribute V individuals	–		–		≤ 40% (35–45)
# Attribute VIt taxa	≤ 2 (1–4)		≤ 2 (1–4)		–
% Attribute VIt individuals	≤ 15% (10–20)		≤ 15% (10–20)		≤ 15% (10–20)
# Mid-water cyprinid taxa	> 0		> 1		> 1
BCG Level 4	**Small**		**Medium**		**Large**
# Attribute I+II+III taxa	> 1 (0–3)		> 1 (0–3)		> 1 (0–3)
% Attribute I+II+III individuals	≥ 5% (3–7)		≥ 10% (7–13)		≥ 10% (7–13)
% Most dominant Attribute Va or VIt individual	≤ 65% (60–70)		≤ 65% (60–70)		≤ 65% (60–70)
BCG Level 5	**Small**		**Medium**		**Large**
# Total taxa	> 4 (3–6)		> 4 (3–6)		> 4 (3–6)
# Total individuals	> 100 (90–110)		> 100 (90–110)		> 100 (90–110)
% Attribute V+VIt taxa	–		≤ 65 (60–70)		≤ 65 (60–70)
% Attribute V+VIt individuals	–		≤ 90 (85–95)		≤ 90 (85–95)

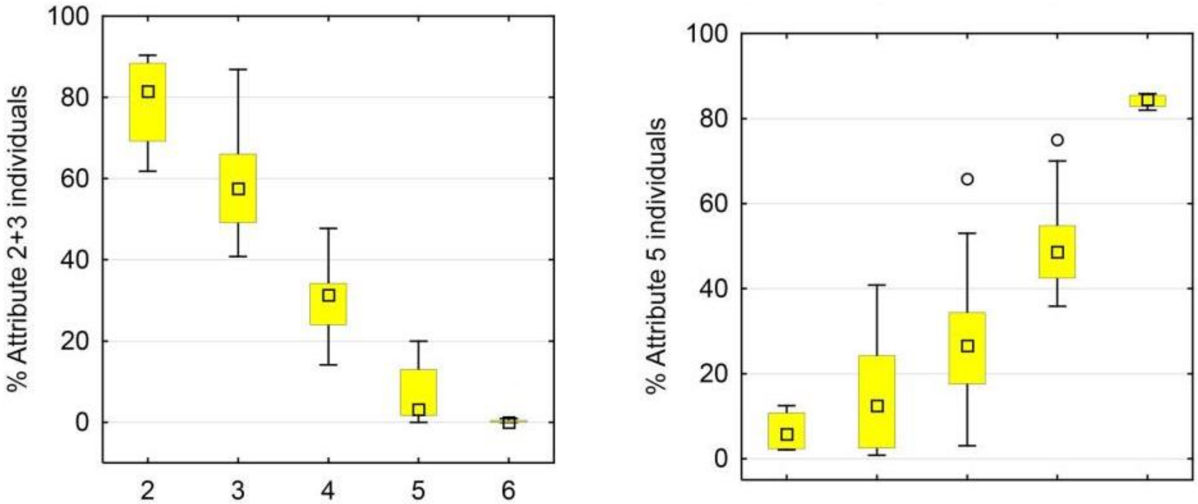


Figure 34. Box plots of sensitive (attribute II+III) and tolerant (attribute V) percent taxa and percent individual metrics for macroinvertebrate calibration samples, grouped by nominal BCG level (expert consensus) (Source: Stamp et al. 2014).

Additional expert panel findings include:

- One headwater site within the TMC watershed (King Spring) was identified as a high quality stream (BCG level 2) with taxa comparable to streams in the adjacent regional park (Little Bennett Regional Park) and with State of Maryland Sentinel Sites for the Piedmont region (Figure 35). Impervious cover for these BCG level 2 sites was at 3% or less. Three other TMC sites with impervious cover ranging between 4% and 11% were rated between BCG levels 3 and 4 (lower condition but considered comparable to "good to fair" conditions). The sites that were approaching BCG level 4 were considered by the experts as candidates for cost effective restoration.
- Sites within the TMC watershed having higher levels of impervious surface were assessed as lower quality. These more degraded sites had elevated levels of specific conductance, an indicator of urban runoff. However, tributaries in excellent to good condition, like King Spring, diluted specific conductance in the lower mainstem TMC.
- Sites within the Piedmont with levels of impervious surface typically higher than 4% showed increasingly degraded aquatic communities. Figure 36 shows average BCG level assignment for benthic macroinvertebrate sampling sites with % sensitive species plotted against % impervious surface. Increased level of impact on the aquatic biota can also be caused by confounding and synergistic effects of other stressors. Additionally, the degree of degradation can be moderated by implementing BMPs. These two considerations likely account for the observed scatter.
- Across Montgomery County both fish and benthic macroinvertebrate assemblages are assessed and may show divergent ratings of condition because of different responses to type and mechanistic pathway of stressors. In some instances, the experts assigned lower condition ratings for the fish community, because there were no or fewer than expected native species. This result was generally attributed to prevention of native fish migration due to dams and other obstacles. Additionally, there was evidence of intrusion of lake fish species from reservoirs so that lake species were dominant over the expected stream species. However, there was sufficient fish habitat and food supply (the benthic macroinvertebrates) to support re-

introduction of native species or migration of other species, such as eel. Depending upon existing temperature regimes, these sites might be excellent sites for re-introduction of native and migratory species.

The decision rules were considered by experts to be applicable to the larger Piedmont region and with minor modification to reflect climate and other latitudinal gradients, useful for assessing biological condition in Piedmont regions in Virginia, Delaware and Pennsylvania.

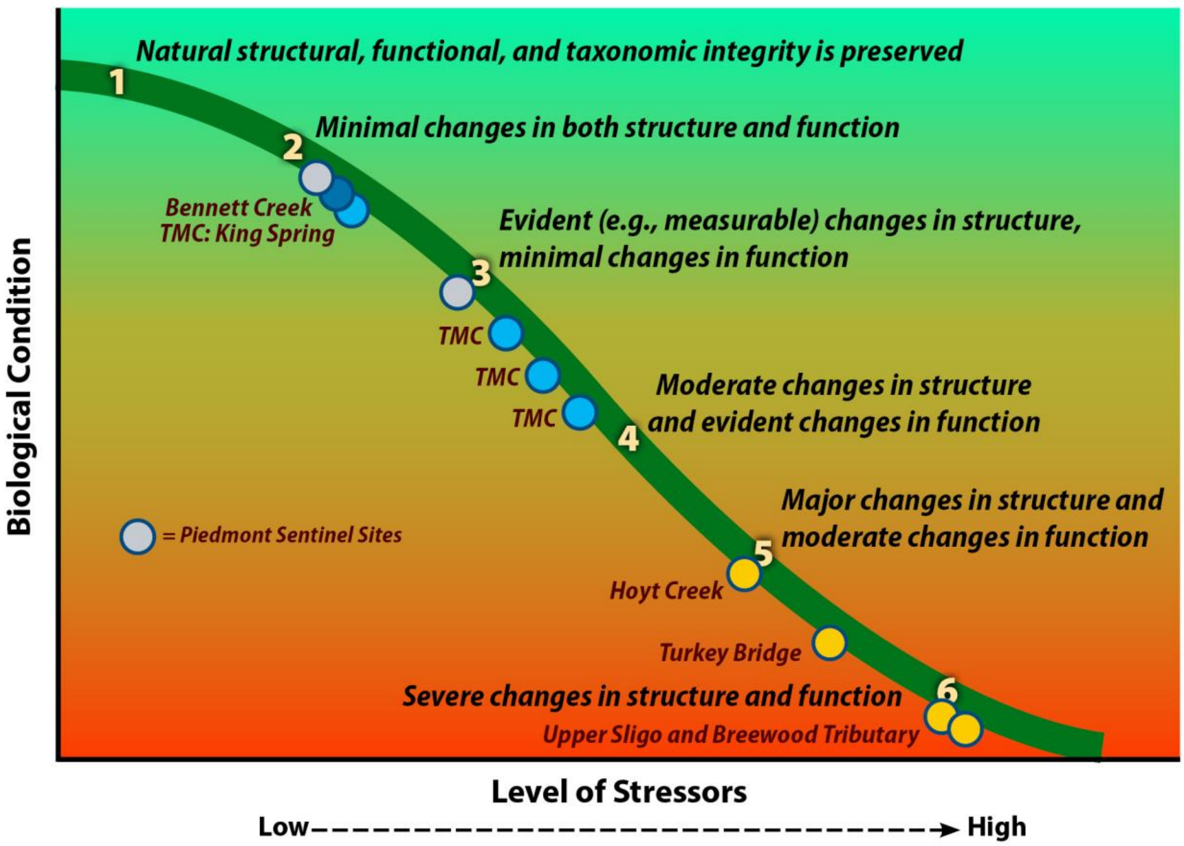


Figure 35. Comparative BCG ratings of macroinvertebrate community data from the county monitoring data set for streams in the TMC watershed and comparable county streams in other watersheds. Data from streams in the State of Maryland Piedmont Sentinel data set were also rated by the experts. The sites were mapped on the gradient according to the expert-derived decision rules for assigning sites to BCG levels.

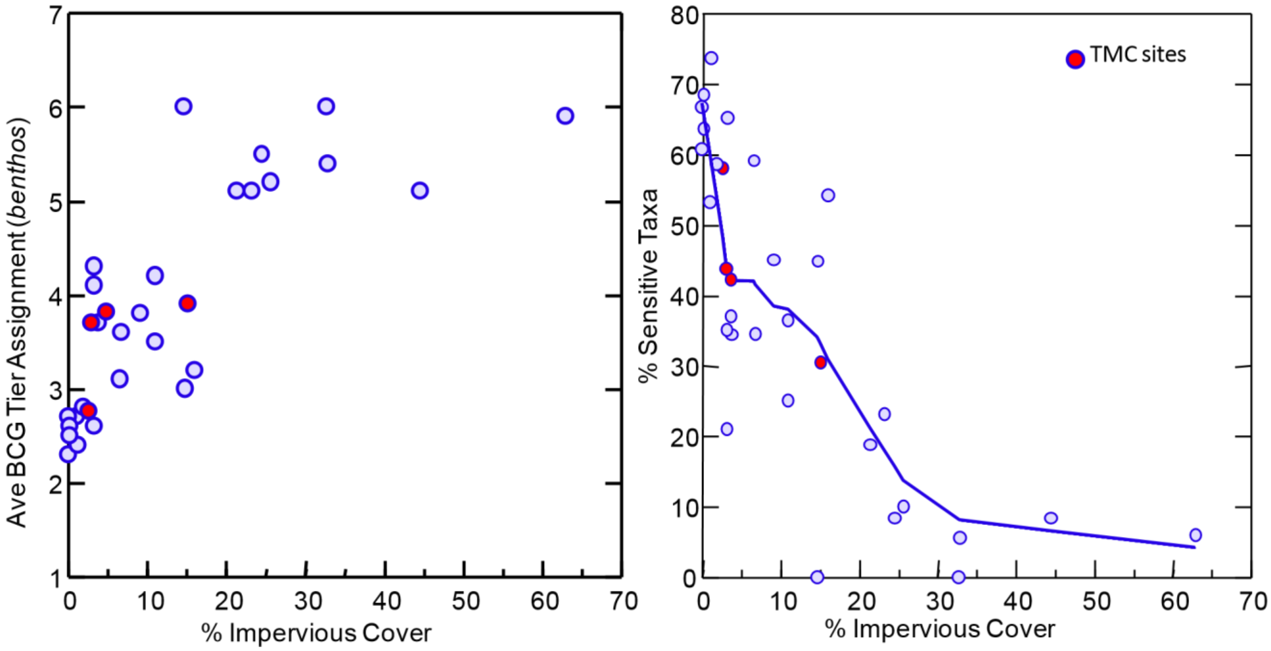


Figure 36. Relationship between average BCG level assignments (left) and % Sensitive Taxa (right) versus % impervious cover. This analysis included sites from throughout the Piedmont Region in Maryland. Ten Mile Creek sites are indicated (red dots).

6.1.4 Use of the Biological Condition Gradient Model in County Planning Decisions

Based on the findings in the environmental analyses associated with the proposed Limited Amendment, the County planning staff and MCDEP scientists concluded that there was significant uncertainty whether high quality aquatic resources assigned special protection, such as TMC subwatershed and streams, would be protected under the 1994 plan. The county planning and MCDEP staff provided several possible development scenarios with predicted outcomes and recommended one option that would modify development in the TMC area while maintaining good environmental conditions (M-NCPPC 2014b). The County Council accepted the recommended option, and it was adopted on April 1, 2014.

The BCG was used in conjunction with expert testimony, peer reviewed literature, research, modeling, and the environmental analysis to inform the County's decision to adopt the 2014 Limited Amendment for Clarksburg. This amendment revised zoning restrictions outlined in the 1994 Master Plan to reduce the impact of development on TMC. The 1994 Master Plan allowed a total impervious cap of 9.8%, while the Limited Amendment proposed a 6.3% impervious surface cap for new development in the most sensitive subwatersheds but allowed a maximum of 15% impervious cover in the Town Center District. The amendment also included a recommendation for increasing forest cover to 65% of the watershed and increasing the size of riparian buffers to better protect the streams and tributaries (M-NCPPC 2014b).

In 2014, the Montgomery County Council adopted the Limited Amendment to the 1994 Clarksburg Master Plan, which focused on TMC. The 2014 Limited Amendment concluded that TMC "warrants extraordinary protection," and offered recommendations for additional zoning restrictions that would allow for continued development, while continuing to study how development and mitigation activities (e.g., implementation of ESD) might affect sensitive water resources in the TMC watershed (M-NCPPC 2014a). The most sensitive streams or tributaries in the TMC system, such as King Spring, are currently at less than 1% impervious cover, so a cap of 6% will likely result in loss of some sensitive species and change from *excellent* to *good,* or potentially *fair*, condition depending on what other development activities occur or protective measures are put in place. For example, the amendment provides for consideration of additional measures (e.g., expanded stream buffer protections) and technology (e.g., ESD) that might minimize these changes (M-NCPPC 2014a). The use of the BCG in conjunction with other data, information, and expert testimony, successfully brought scientific information into the decision-making process and provided for informed decision making that balanced multiple public and private concerns and priorities.

6.1.5 Lessons Learned

Montgomery County found that the BCG framework was a good tool to better articulate current conditions in TMC and illustrate how water quality could be impacted by future development as outlined in the 1994 Master Plan. The 2014 Limited Amendment will allow for continued development with some restrictions on impervious cover. Because the BCG can be used in conjunction with monitoring data to detect incremental changes in stream health, county scientists will be able to closely monitor the effects of using ESD and other BMPs to mitigate the impacts of development on sensitive waters. County officials found that the BCG gave experts and the public a common understanding of water quality issues and allowed for informed policy making.

In the future, the County plans to use the BCG as an interpretative framework to examine restored sites and identify incremental improvements or declines in biological condition. Future use of this information might also include using county data for restoration modeling. In addition, the BCG might be used as one way to reconcile databases maintained at the County-level with those at the state level. Ultimately, one goal of such an effort could be to have county-level data used by the state when classifying streams.

6.2 Pennsylvania: Using Complementary Methods to Assess Biological Condition of Streams

6.2.1 Key message

Pennsylvania Department of Environmental Protection (PA DEP) implements a multi-tiered benchmark decision process for assessing attainment of ALU for wadeable, freestone, riffle-run streams in Pennsylvania. This multi-tiered approach incorporates stream size and sampling season as factors for determining ALU attainment based on benthic macroinvertebrate sampling. A BCG calibrated for freestone, riffle-run streams is used to supplement the state's primary screening tool, the IBI for benthic macroinvertebrates (PA DEP 2013a).

6.2.2 Using Index of Biological Integrity to Assess Aquatic Life Uses

PA DEP has developed a multimetric benthic macroinvertebrate IBI for the wadeable, high gradient, freestone[26] streams in Pennsylvania using the reference condition approach (PA DEP 2012). These streams are non-calcareous, or softwater, free flowing streams and comprise the majority of the state's streams. PA DEP has alternative assessment methods in place for other stream types (i.e., low-gradient pool-gliders, karst- [limestone]-dominated). The IBI provides an integrated measure of the overall condition of a benthic macroinvertebrate community in a water body by combining multiple metrics into a single index value. A number of different metric combinations were evaluated during IBI development. Based on discrimination efficiencies, correlation matrix analyses, and other index performance characteristics, PA DEP selected the following six metrics for inclusion as core metrics in the MMI (PA DEP 2012):

1. **Total Taxa Richness—**This taxonomic richness metric is a count of the total number of taxa in a subsample. Generally, this metric is expected to decrease with increasing anthropogenic stress to a stream ecosystem, reflecting loss of taxa and increasing dominance of a few pollution-tolerant taxa.
2. **Ephemeroptera + Plecoptera + Trichoptera Taxa Richness (EPT)—**This taxonomic richness metric is a count of the number of taxa belonging to the orders Ephemeroptera, Plecoptera, and Trichoptera in a sub-sample—common names for these orders are mayflies, stoneflies, and caddisflies, respectively. The aquatic life stages of these three insect orders are generally considered sensitive to, or intolerant of, many types of pollution (Lenat and Penrose 1996), although sensitivity to different types of pollution varies among specific taxa in these insect orders. This metric is expected to decrease in value with increasing anthropogenic stress to a stream ecosystem, reflecting the loss of taxa from these largely pollution-sensitive orders.
3. **Beck's Index—**This taxonomic richness and tolerance metric is a weighted count of taxa. The name and conceptual basis of this metric are derived from the water quality work of William H. Beck in Florida (Beck 1955). This metric is expected to decrease in value with increasing anthropogenic stress to a stream ecosystem, reflecting the loss of pollution-sensitive taxa.
4. **Shannon Diversity—**This community composition metric measures taxonomic richness and evenness of individuals across taxa in a sub-sample. This metric is expected to decrease in value with increasing anthropogenic stress to a stream ecosystem, reflecting loss of pollution-sensitive

[26] Freestone is a term familiar to fly-fisherman, denoting streams with little groundwater influence showing high annual variation in flow (spring freshet, summer drought).

taxa and increasing dominance of a few pollution-tolerant taxa. The name and conceptual basis for this metric are derived from the information theory work of Claude Elwood Shannon (Shannon 1948).

5. **Hilsenhoff Biotic Index—**This community composition and tolerance metric is calculated as an average of the number of individuals in a sub-sample, weighted by pollution tolerance values. Developed by William Hilsenhoff, the Hilsenhoff Biotic Index (Hilsenhoff 1977, 1987a, 1987b, 1988; Klemm et al. 1990) generally increases with increasing ecosystem stress, reflecting increasing dominance of pollution-tolerant organisms.
6. **Percent Sensitive Individuals—**This community composition and tolerance metric is the percentage of individuals in a sub-sample and is expected to decrease in value with increasing anthropogenic stress to a stream ecosystem, reflecting loss of pollution sensitive organisms.

PA DEP determined that these six metrics all exhibited a strong ability to distinguish between reference and stressed conditions in testing with benthic invertebrate assemblage data from riffle run habitats in wadeable, freestone streams. When used together in an MMI, these metrics provide PA DEP with a consistent and defensible index for assessing the biological condition of these streams (PA DEP 2012).

6.2.3 Use of the Biological Condition Gradient to Complement Aquatic Life Use Assessments

PA DEP is exploring use of a BCG to describe the biological characteristics of wadeable, freestone streams along a gradient of stress. More than 75% of Pennsylvania is in the hills and low mountains of the Appalachian Highlands, so streams throughout the state are predominantly relatively high gradient (> 1% slope) (Figure 37 and Figure 38). Pennsylvania is largely forested, but there are significant areas where agricultural land use, including row-crops and pasture, is dominant (Figure 39). Limestone and spring-dominated streams occur in parts of southeast, south-central and east-central Pennsylvania. The BCG assessments and model discussed in the case study do not apply to this subset of streams.

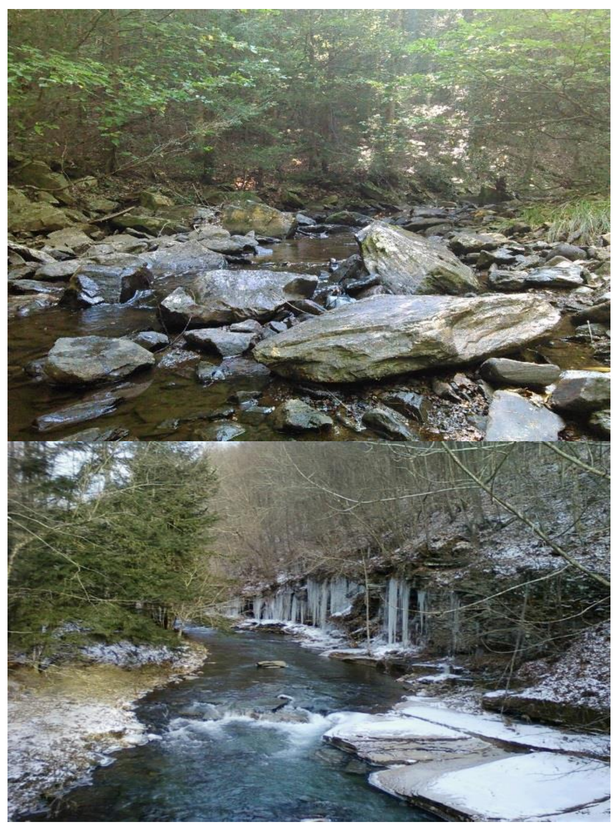


Figure 37. Top: Carbaugh Run, Adams County; Bottom: Rock Run, Lycoming County (Photos courtesy of PA DEP).

Between 2006 and 2008, PA DEP conducted a series of expert workshops to calibrate a BCG along a gradient from minimally to heavily stressed conditions (PA DEP 2013b). To develop the BCG for the wadeable, freestone streams, biologists from PA DEP, in conjunction with external taxonomic experts and scientists (e.g., the Delaware River Basin Commission, Western Pennsylvania Conservancy, and EPA), used the BCG attributes that characterize specific changes in community taxonomic composition (PA DEP 2013b). For example, in the highest levels of the BCG, locally endemic, native, and sensitive taxa are well represented, and the relative abundances of pollution-tolerant organisms are typically lower. With increasing stress, more pollution-tolerant species may be found with concurrent loss of pollution-sensitive species. At the beginning of the expert workshop, the participants assigned a BCG attribute for sensitivity to stress (i.e., attributes I–V) to each macroinvertebrate taxon based on expert knowledge and biological response data. The data used was from sampling sites that spanned a range of condition from reference quality (e.g., at or close to minimally disturbed conditions) to heavily stressed sites (PA DEP 2013b). Using the BCG level descriptions of predicted changes in the attributes as a guide, the expert panel then assigned each site to one of the six BCG levels and developed candidate decision rules (Figure 40, Table 31).

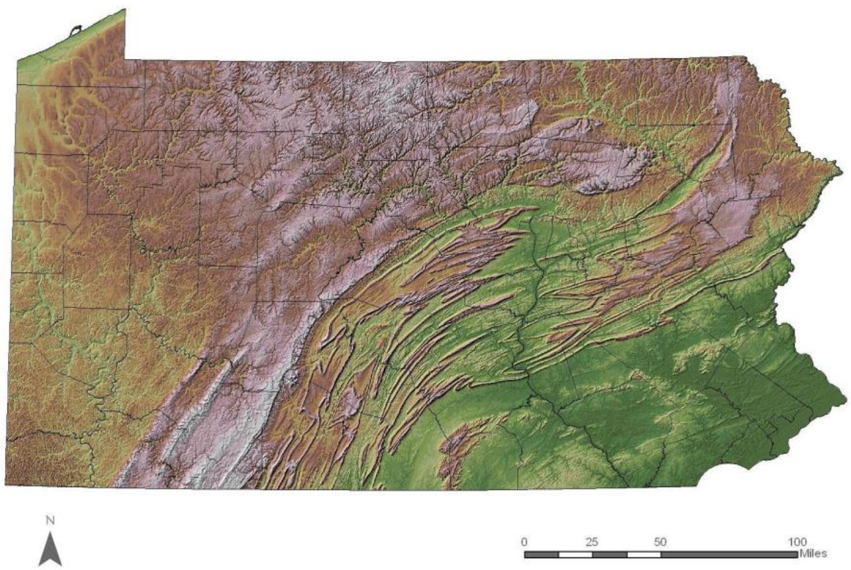


Figure 38. Topographic Map of Pennsylvania.

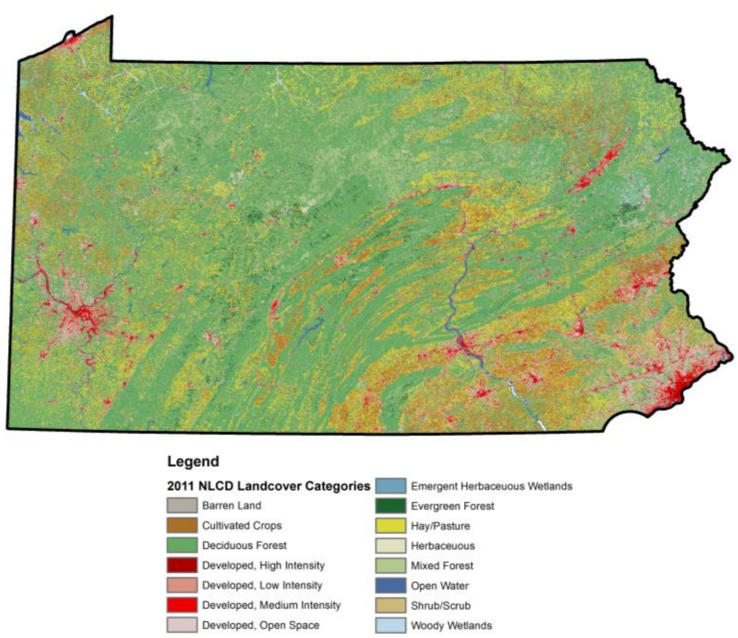


Figure 39. Pennsylvania Land Use.

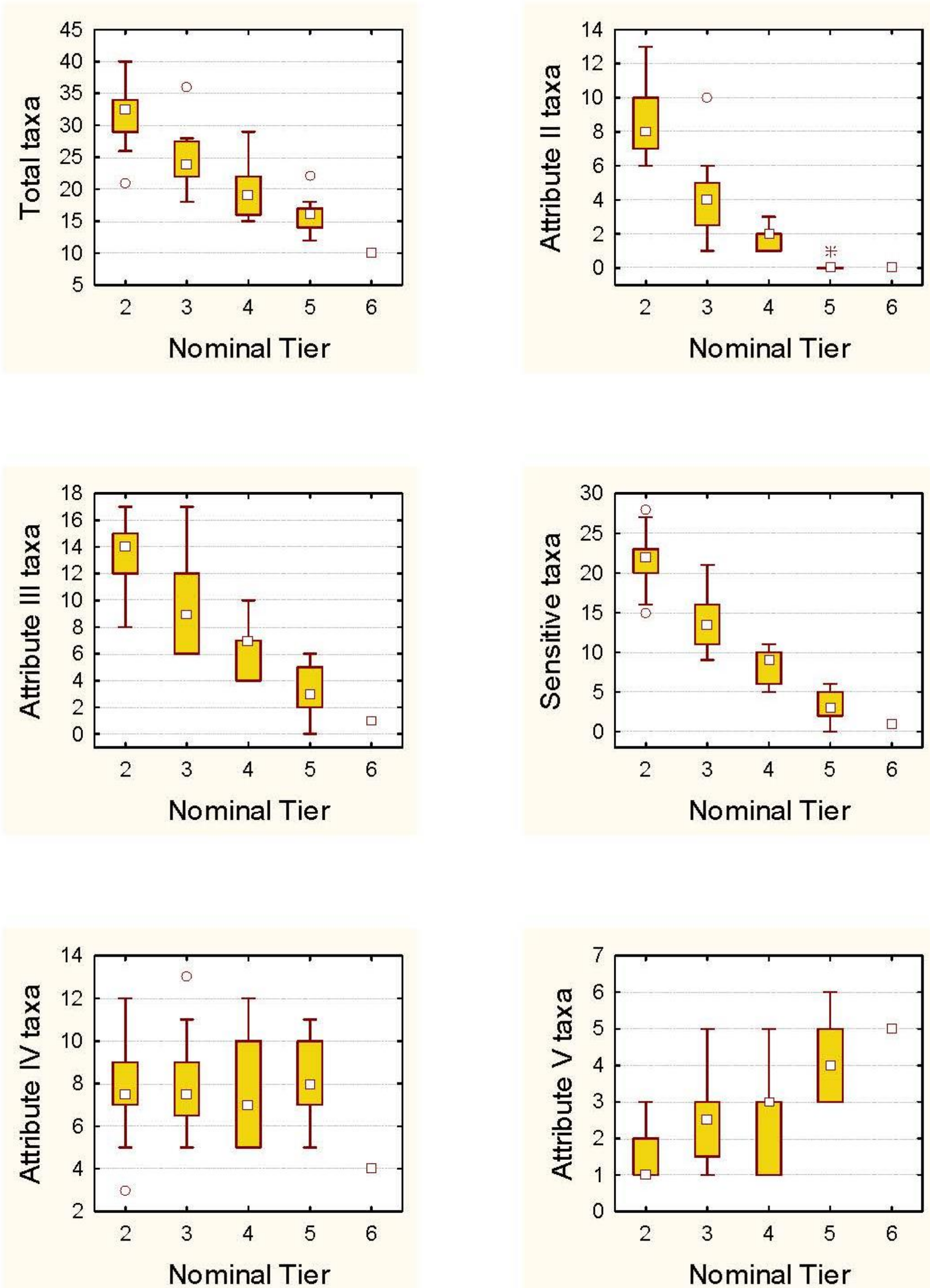


Figure 40. Box plots of BCG metrics, by nominal level (group majority choice). Sensitive taxa are the sum of both attribute II (highly sensitive) and attribute III taxa (intermediate sensitive) (Source: Gerritsen and Jessup 2007c).

Table 31. Potential narrative decision rules for invertebrate samples from Pennsylvania high gradient streams (modified from Gerritsen and Jessup 2007c)

Attributes	BCG Level				
	2	3	4	5	6
All Taxa	> 25 taxa	> 20 taxa		≥ 10 taxa No single taxon ≥ 50% of abundance ≥ 50 individuals in sample	Low richness or low abundance
I. Historically documented, sensitive, long-lived, or regionally endemic taxa	*No rules determined for attribute I*				
II. Highly sensitive taxa	Taxa II ≥ 33% of Taxa III	Taxa II present (> 0)	May be absent (no rule)		
III. Intermediate sensitive taxa	Taxa (II + III) ≥ 50% of all taxa Indiv (II + III) ≥ 50% of all indiv	Taxa (II + III) ≥ 30% of all taxa Indiv (II + III) ≥ 30% of all indiv	Taxa (II + III) present (≥ 10% of taxa, or 2 taxa) Indiv (II + III) ≥ 15%–20% of all indiv		
IV. Intermediate tolerant taxa	*No rules determined for attribute IV*				
V. Tolerant taxa	Few tolerant taxa; Tolerants are small % of total abundance (≤ 5%)	Tolerant individuals ≤ 20% of total abundance	Tolerant individuals ≤ 40% of total abundance		Tolerant individuals may dominate
Indicator taxa	Many EPT taxa; EPT ≥ 15	Tolerant Caddisflies ≤ 20% abundance EPT ≥ 12	Tolerant Caddisflies ≤ 40% abundance EPT ≥ 8	Tube worms not dominant; ≤ 50% of abundance	Mayflies may be absent; Tube worms may dominate

Each sampling site used to develop and test the BCG decision rules had corresponding IBI scores. The IBI uses metrics that are similar in objective to the BCG attributes, but which are calculated differently (PA DEP 2013a). The total IBI score is based on the sum or average of all metrics, while BCG decision rules are based on specific attribute groups and patterns of change along a gradient of stress (e.g., attributes II and III for the higher levels and attribute V for lower levels).

For all the evaluated samples, PA DEP biologists analyzed the relationship between a sample's BCG level assignment with its corresponding IBI score (PADEP 2013b). A strong correlation existed between the calibrated BCG level assignments and the IBI scores (Figure 41). On the basis of this comparative analysis, PA DEP determined that with further testing and evaluation, the IBI scores could potentially be used to discriminate BCG levels. PA DEP is evaluating using the BCG to describe the biological characteristics of streams assessed based on the IBI scores; for example, the reference sites clustered at IBI scores near 80 and above would be interpreted as primarily comparable to BCG levels 1–2. On the basis of taxonomic information, and without knowledge of the IBI scores, the experts assigned these sites to BCG levels 1.5 to 2.5. BCG level 2 represents close to natural conditions (e.g., minimal changes in structure and function relative to natural conditions; supports reproducing populations of native species of fish and benthic macroinvertebrates). This information can meaningfully convey to the public the biological characteristics of waters in the context of the CWA and the goal to protect aquatic life. PA DEP is evaluating use of the BCG to complement the IBI in assessing ALU attainment and to help identify potential high-quality (HQ) or exceptional value (EV) streams. As a first step in application of the BCG, PA DEP has incorporated BCG attributes for taxa sensitivity to stress as part of its protocol for wadeable, freestone streams (Figure 42) (PA DEP 2013a).

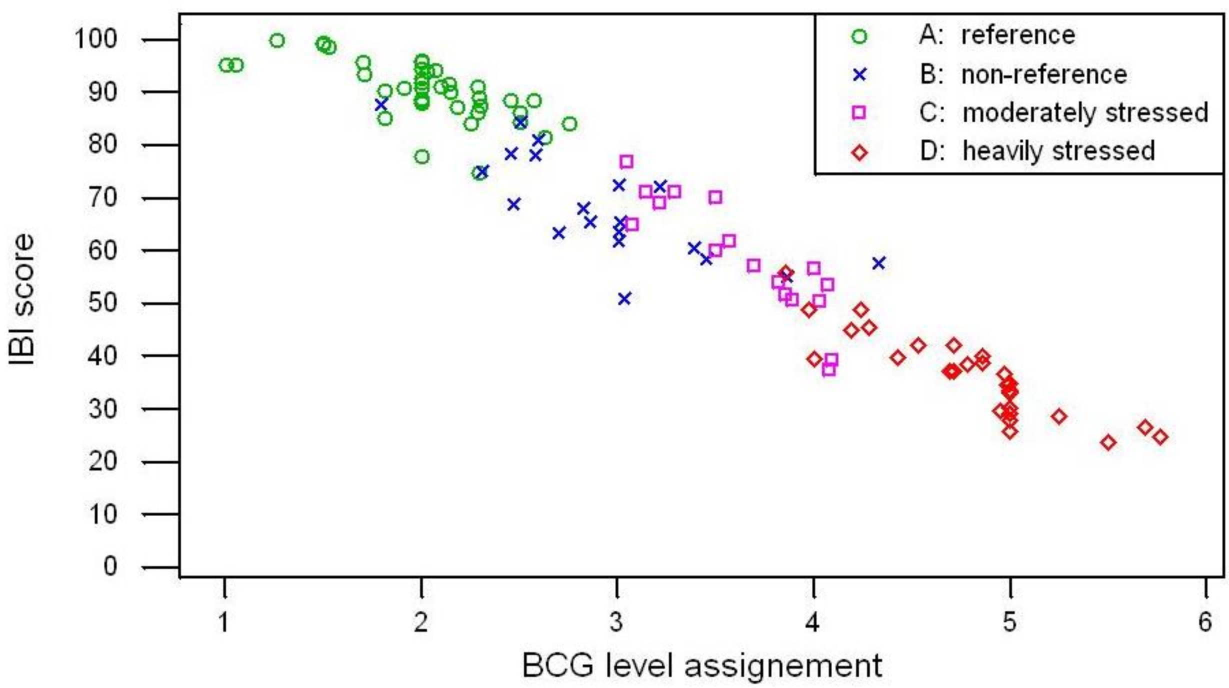


Figure 41. Comparison of calibrated BCG level assignments (mean value) and IBI scores for freestone streams representing range of conditions from minimal to severely stressed.

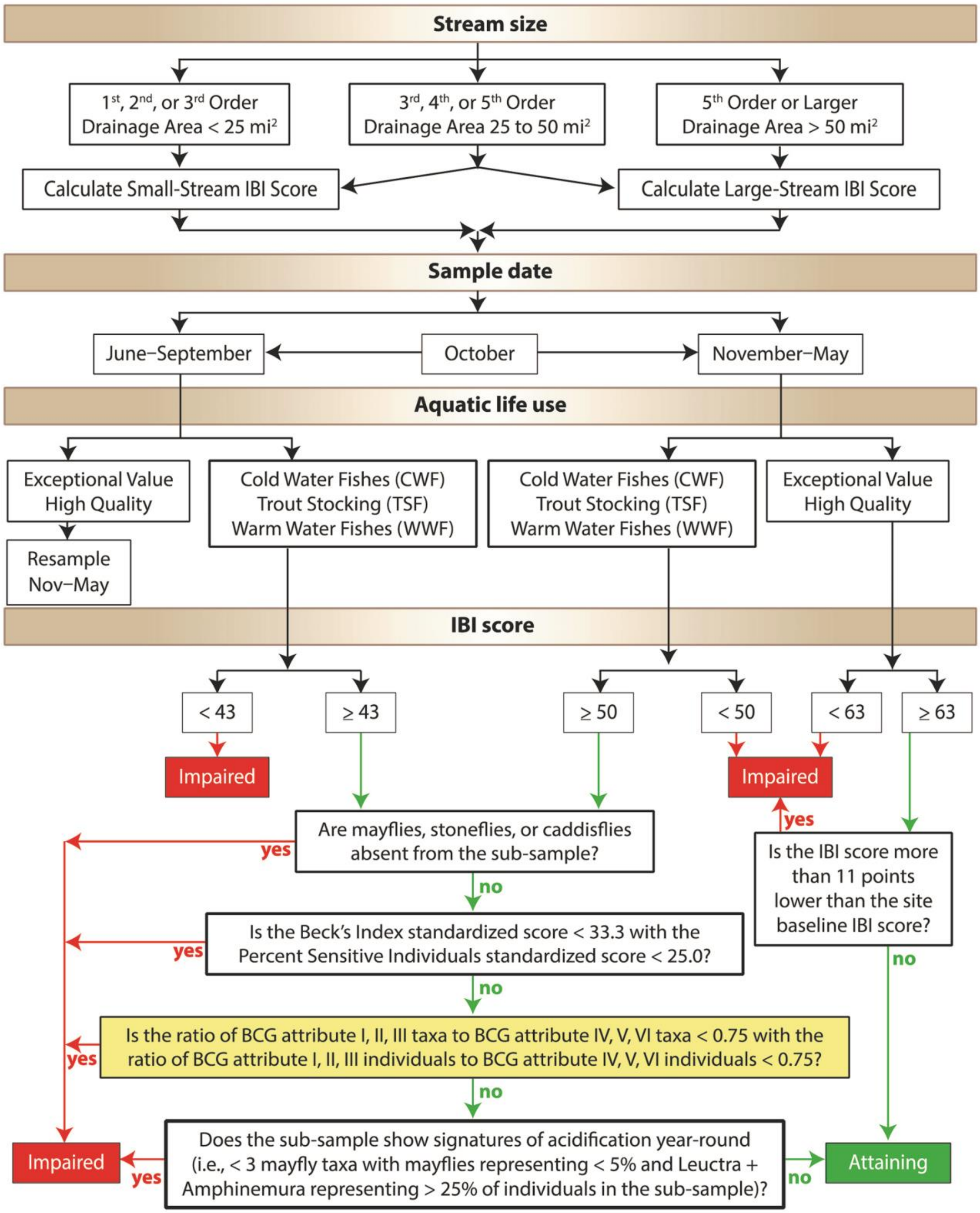


Figure 42. Multi-tiered benchmark decision process for wadeable, freestone, riffle-run streams in Pennsylvania (Modified from PA DEP 2013a). The ratio of BCG attributes for sensitive to tolerate taxa (i.e., attributes I, II, and III to attributes IV, V, and VI) are included as part of attainment determination (see yellow box). Rules have not been defined for attribute I and IV but these attributes are included in the assessment protocol if decision rules are developed in the future and determined to be appropriate to include.

6.2.4 Potential Application to Support Aquatic Life Use Assessments and Protection of High Quality Waters

Pennsylvania's regulations define waters of EV that are of unique ecological or geological significance. EV streams are given the highest level of protection and constitute a valuable subset of Pennsylvania's aquatic resources. To support protection of these waters, PA DEP is considering the use of a discriminant analysis model to evaluate the relationship between condition of the watershed, a stream, and its aquatic biota (e.g., the connection of riparian areas with a stream and the floodplain or the spatial extent of stressors and their sources in the watershed). PA DEP is evaluating the use of a discriminant model that incorporates measures of land use and physical habitat along with IBI scores and the BCG to make distinctions between EV and HQ waters. PA DEP is also evaluating how to consider effects of habitat fragmentation and spatial and temporal extent of stress. The results of this effort could potentially support state water quality management decisions on where to target resources for sustainable, cost-effective protection of EV waters and healthy watersheds. Through this work, PA DEP can provide EPA valuable feedback on the technical development and potential program application for BCG with specific focus on defining indicators for BCG attributes IX (spatial and temporal extent of detrimental effects) and X (ecosystem connectance).

6.3 Alabama: Using the Biological Condition Gradient to Communicate with the Public and Inform Management Decisions

6.3.1 Key Message

ADEM has strategically built a comprehensive biological monitoring program over the past four decades and has, more recently, invested in developing BCGs for streams in all regions of the state. ADEM has identified reference conditions in order to better characterize current water quality condition, and it has built increasing capability in terms of data management. As ADEM's capabilities have evolved, it is applying biological data, biological indices, and the BCG for a variety of management purposes, including identification of high quality waters and waters that need restoration. As part of this process, ADEM has improved its ability to communicate to the public on the condition of streams and to measure incremental improvements in condition. Though the state is developing and applying the BCG and biological assessments on a statewide basis, this case study reports on the development and application of a BCG for the high gradient streams of Northern Alabama.

6.3.2 Program Development

Since 1974, ADEM has been monitoring its surface water quality, and the capabilities of the monitoring program have evolved over time. In 1997, ADEM first formalized a coordinated monitoring strategy to outline its surface water quality monitoring efforts. Today, ADEM collects biological, chemical, and physical data and uses those data to inform management decisions, including assessing the health of state waters, determining whether those waters are meeting their designated uses, and identifying impacts from a variety of sources (ADEM 2012).

ADEM continues to build its monitoring program to meet emerging data needs, and it is currently evaluating the use of its biological data in new ways. ADEM conducted a preliminary critical elements review[27] of its biological assessment program in 2006 to assess the strengths of the technical program. The review highlighted ADEM's efforts to that point, and it included recommendations for enhancements relative to design, methodology, and execution for credible data as a basis of making informed decisions regarding the ecological condition of Alabama's streams. The review resulted in a recommendation that ADEM fully implement its monitoring strategy to accomplish a variety of goals, including more complete development of reference conditions and site criteria, and development and/or refinement of macroinvertebrate, fish, and diatom community assessment methods; ecological attributes; response patterns; and indices along a continuous BCG scale. The review also highlighted the need for an improved and enhanced database management system; improved technical capabilities to carryout survey needs; statewide completion of monitoring unit delineation; and incorporation of up-to-date land cover data sources.

Since the 2006 review, ADEM has continued to make improvements in the technical capabilities of its biological assessment program. In 2008, ADEM used data collected in 1994–2005 to develop MMIs for high and low gradient streams. The indices were used for site assessments with thresholds derived from the reference distribution. At the same time, the biological database was updated to a new platform, integrated into ADEM's centralized surface water database, Alabama Water-Quality Assessment and

[27] For more information about Critical Element Review, see *Biological Assessment Program Review: Assessing Level of Technical Rigor to Support Water Quality Management* (USEPA 2013, http://www.epa.gov/wqc/biological-assessment-technical-assistance-documents-states-and-tribes. Accessed February 2016.)

Monitoring Data Repository (ALAWADR), which houses chemical, physical, and biological data. In 2009, the database was modified to calculate macroinvertebrate metrics and indices. Incorporating these tools into ALAWADR assisted greatly in the development and testing of ecological attributes, stress-biological response patterns, and indices along continuous BCG and stressor scales. In 2013, ADEM expanded the effort to use data from the 1994–2011 period to incorporate additional reference site data to refine the site classes, and MMIs (Jessup 2013). In these efforts, ADEM considered regional differences in biological habitat and species distribution, and it found that variability was best explained using ecoregions[28] for classification. ADEM calibrated the MMIs to categorize water quality on a scale from *Very Good* to *Very Poor* (Jessup 2013).

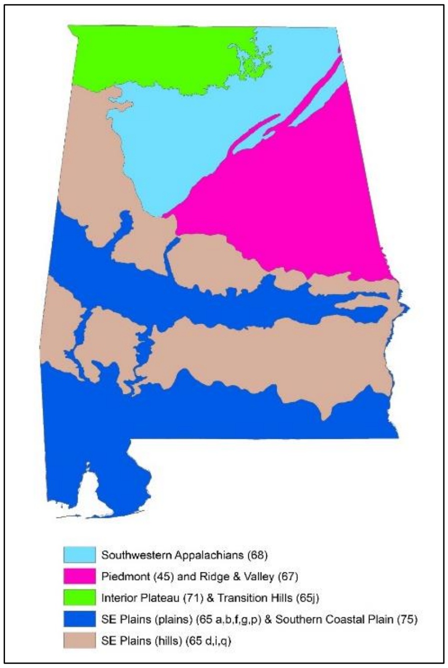
In a similar effort spear-headed by the Geological Survey of Alabama, ADEM and the Alabama Department of Conservation and Natural Resources collaborated to develop statewide multimetric fish community indices. In 2004, the Geological Survey of Alabama completed refinement of collection methods developed by the Tennessee Valley Authority and established five site classes, or ichthyoregions, primarily based on ecoregions and basins. Statewide MMIs were completed in 2011–2012.

6.3.3 Index Development

As a result of the work and collaboration among state agencies discussed above, ADEM developed biological indices for both macroinvertebrates and fish statewide. Assessment thresholds were established for both assemblages using similar analytical methods though there were differences in site classification and threshold delineation. First, similar regions for classification were identified for each assemblage, but they were not identical (Figure 43). For site classification, the similarity of species composition relative to ecoregions, drainage basins, and other natural site characteristics was analyzed. Shared environmental variables associated with the ecoregional distinctions for both assemblages included elevation, temperature, and percent cobble and boulder substrate. However, differences in classification for the two assemblages were attributed to the dependence on drainage continuity for fish migrations, whereas macroinvertebrates (especially insects) can move among drainages by flying during adult stages.

Second, for benthic macroinvertebrates, candidate reference sites were identified based on measurements of disturbances both at the site and in the landscape. A watershed disturbance gradient (WDG) was calculated using land use coverage (e.g., percent urban, row crop, and/or pasture in the catchment) and road density (Brown and Vivas 2005; ADEM 2005). Figure 44 shows broad land cover patterns throughout the state. The 25th percentile of the WDG was used as the threshold for selecting candidate reference streams. These reference streams experienced minimal to moderate levels of stress and are considered "least disturbed" conditions (Stoddard et al. 2006). However, for some regions, land use intensity as measured by the WDG was considerably higher and more widespread, reflecting regional patterns in agricultural and urban land use. Reference streams in the regions with more intensive development (e.g., higher WDG scores) generally had lower biological scores (e.g., benthic macroinvertebrate scores) (Table 32). Figure 45 shows the range of land intensity scores in sites assessed by ADEM, including reference sites.

[28] Ecoregions describe areas with similar features related to geology, physiography, vegetation, climate, soils, land use, wildlife, and hydrology.



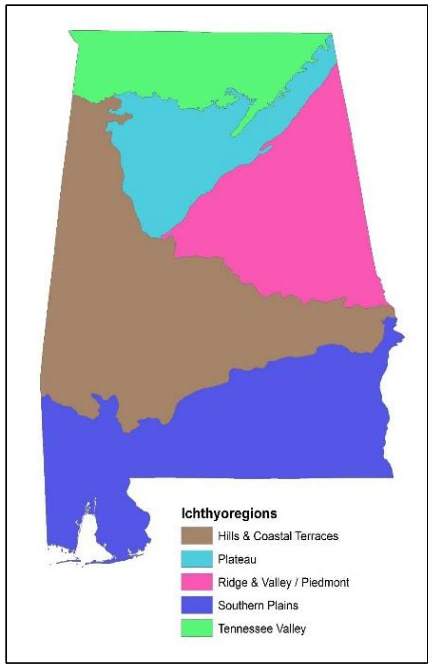


Figure 43. Left: Macroinvertebrate site classes in Alabama; Right: Fish site classes in Alabama.

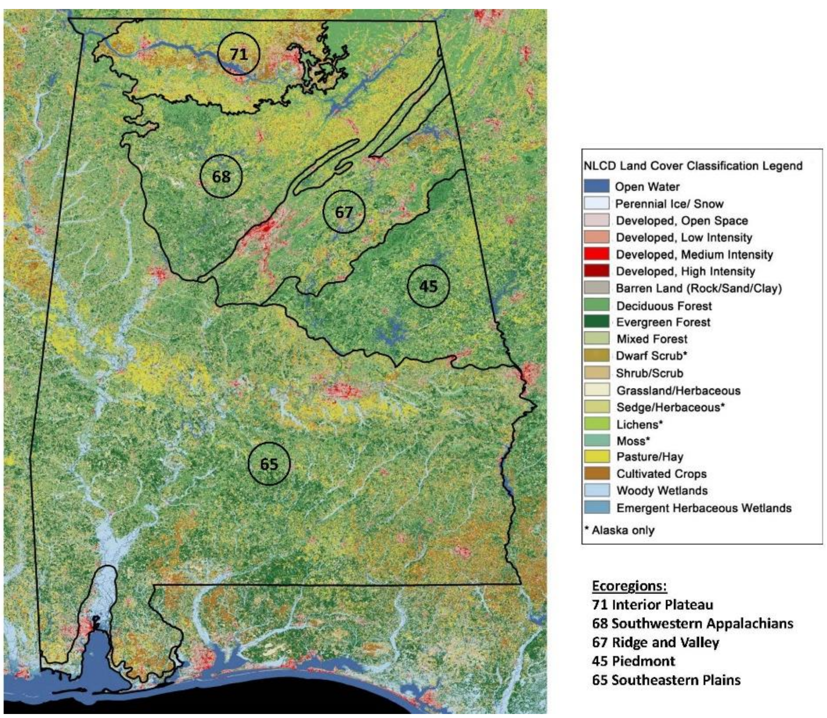


Figure 44. Alabama land use/land cover map.

Table 32. Characterization of Reference Conditions Using WDG and the Alabama Macroinvertebrate MMI for streams. WDG scores increase with level of land use activity.

Macroinvertebrate Site Class	Median Reference WDG Score	Benthic Macroinvertebrate MMI Score: 25th Quantile of Reference

Macroinvertebrate Site Class	Median Reference WDG Score	Benthic Macroinvertebrate MMI Score: 25th Quantile of Reference
Interior Plateau	61	43
Southeastern Plains–Hills	64	47
Piedmont, Ridge & Valley	46	69
Southwest Appalachians	31	58
Southeastern Plains–Plains	90	45

Additionally, there are differences in how the two assemblage indices were scored and benchmarks established. As described above, the benchmark for the macroinvertebrate index was based on a reference condition approach. Reference sites were selected based on abiotic parameters that met predetermined selection criteria and a 25% threshold was established (Table 32). However, for fish, the range of index scores from all sites was divided into five condition categories: excellent, good, fair, poor, and very poor (Figure 46). The thresholds between fish categories were selected to create a balanced distribution of conditions among the sampled sites, with most samples in the fair category, and similar numbers of excellent and good samples compared to poor and very poor samples (Figure 46; O'Neil and Shephard 2011). Thus, the reference condition for macroinvertebrates and the excellent and good categories for fish are not a one for one match. ADEM wanted to develop the BCG model and numeric decision rules so that benthic macroinvertebrate and fish assemblage data could be mapped on the BCG and interpreted against a uniform standard despite differences in sample collection and analysis.

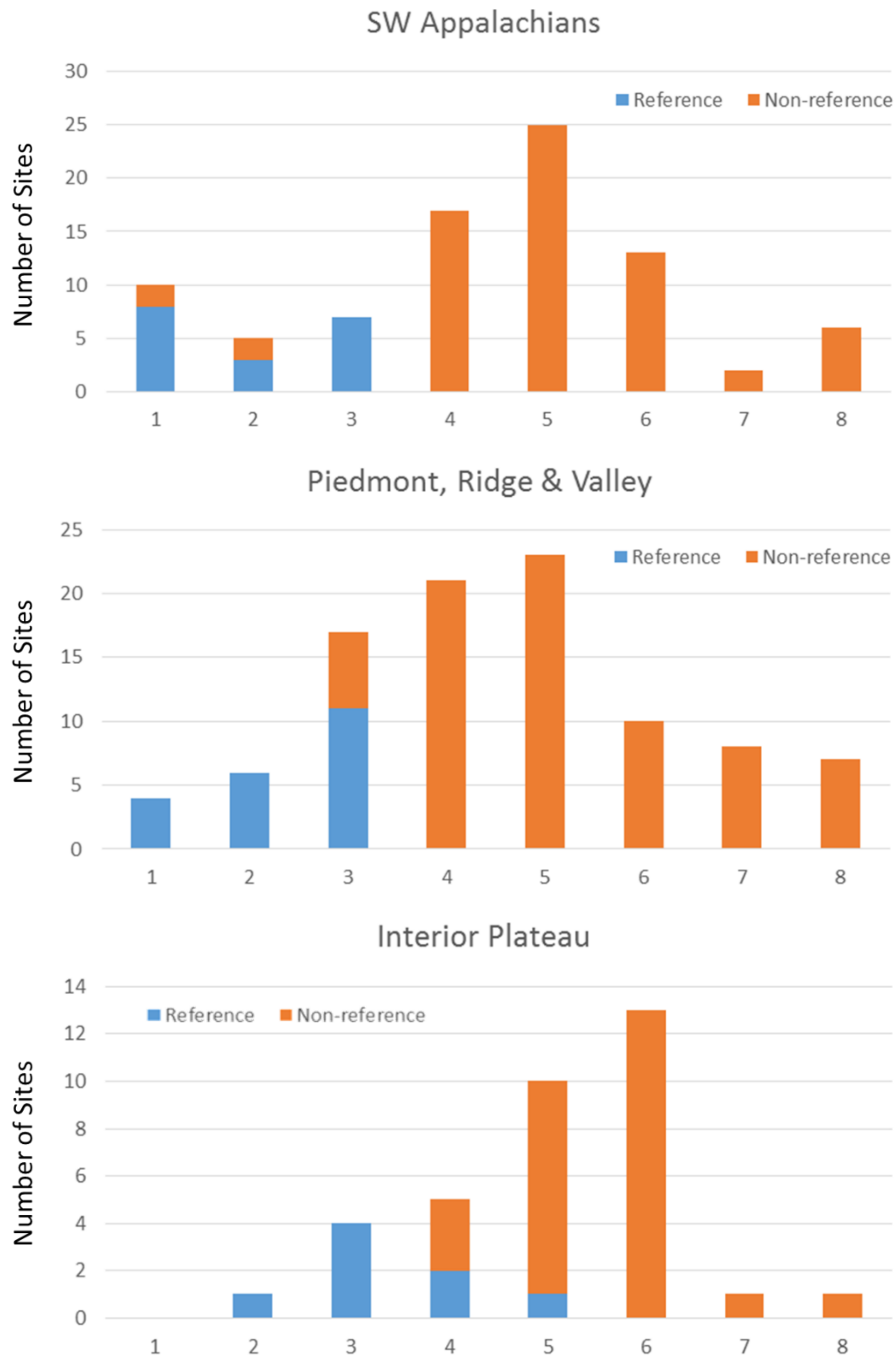


Figure 45. Frequencies of sites in ranked WDG categories (x-axis), distinguishing reference and non-reference sites in each site class. Distributions are based on sites monitored in ADEM's biological assessment program. WDG categories are numerically ranked with increased levels of stress. ADEM converted the WDG scores to ranks 1–8, with lower numbers representing less disturbance.

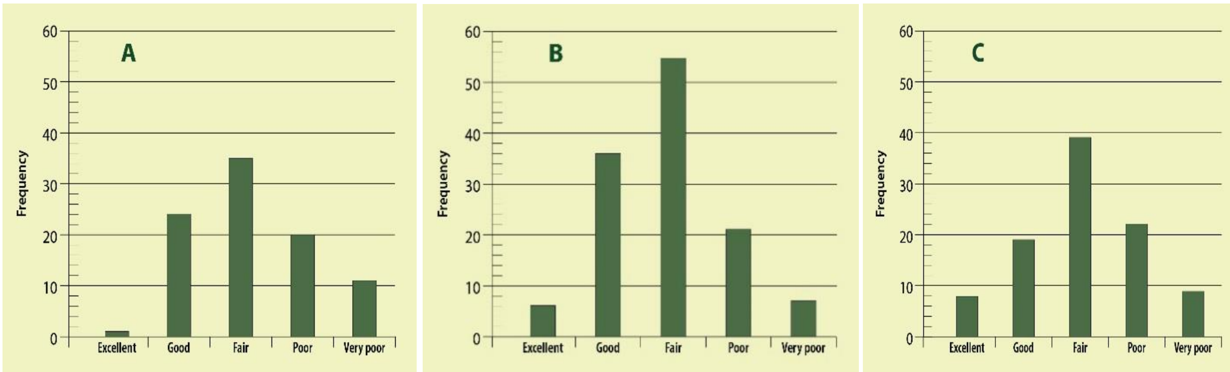


Figure 46. Frequency distribution of fish IBI condition categories for sites in the three ichthyoregions discussed in this case study: the (A) Plateau; (B) Piedmont, Ridge, and Valley; and (C) Tennessee Valley site classes. The x-axis is divided into five condition categories: excellent, good, fair, poor, and very poor.

6.3.4 The Biological Condition Gradient

In 2014, ADEM and Geological Survey of Alabama convened an expert panel of scientists from the state, outside agencies, academia, and other research organizations. The charge to the expert panel was to develop a quantitative BCG and to use the BCG to calibrate BCG-based indices for fish and macroinvertebrate assemblages for wadeable streams in Alabama. The first phase of BCG development was on wadeable streams in Northern Alabama in three ecoregions: the Interior Plateau, Southwest Appalachian, and the Piedmont Ridge Valley ecoregions. This case study reports on these results. The second phase of BCG development is underway for the coastal plain streams in central and southern Alabama.

Wadeable streams in northern Alabama are higher gradient relative to streams in the coastal plains of southern Alabama and tend to have a riffle habitat (Figure 47). Experts developed numeric decision rules to predict the BCG level of a stream based on site classes for fish and macroinvertebrates (Jessup and Gerritsen 2014). Models were then developed to replicate the expert decisions for assigning new samples to BCG levels 2–6 without having to reconvene the expert panel. There were no sites in the data set used to develop the BCG that the experts considered comparable to BCG level 1 (undisturbed), so the experts conceptually described the expected biological community for a BCG level 1. The conceptual description provided a shared, narrative starting point for assessing incremental changes from BCG level 1 to BCG level 6. The final modeled BCG levels correctly predicted the expert ratings of actual site data for BCG levels 2–6 in 94% and 96% of cases for macroinvertebrates and fish, respectively.

Figure 47. Example of range in typical northern Alabama streams with riffle-run habitat. Top: Hendriks Mill Branch; Bottom: Hatchet Creek.

As the first step in developing the BCG model for northern Alabama streams, the benthic macroinvertebrate and fish species were assigned BCG attributes corresponding to their prevalence and sensitivity to disturbance. These characteristics were analyzed using abundance of individuals and general additive models (GAMs) based on the capture probability of each taxon along the WDG scale. Experts in the workgroup used the model results and their own experience to assign attributes to each taxon. Taxa with steeply descending model slopes were sensitive to disturbance and were assigned attributes II or III (e.g., highly and intermediate pollution sensitivity) based on the slope of the response curve (e.g., capture probability) (e.g., *Acroneuria* in Figure 48). Taxa with flat slopes were found in a variety of disturbance conditions and were assigned to BCG attribute IV (taxa of intermediate tolerance). Taxa with increasing capture probabilities with increasing disturbance were assigned to BCG attribute V (tolerant taxa) (e.g., *Ferrissia* in Figure 48). In the second step of the BCG process, the experts used the attribute assignments in developing the decision rules for assigning sites to BCG levels (Table 33).

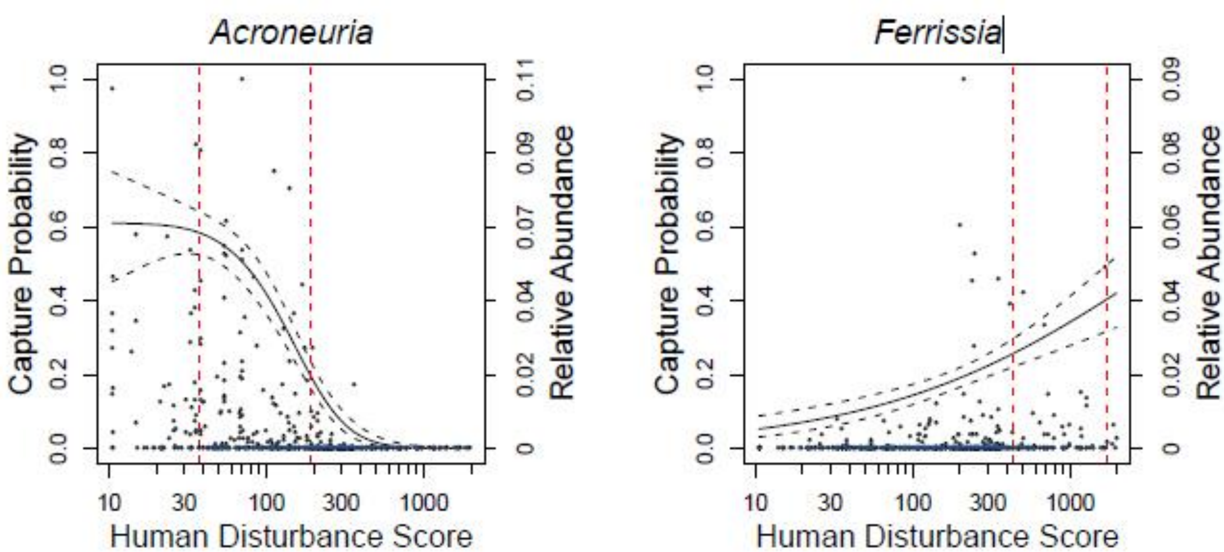


Figure 48. Taxa relative abundance and the GAM slope based on capture probabilities for *Acroneuria* (Plecoptera: Perlidae; attribute III) and *Ferrissia* (Gastropoda: Ancylidae; attribute V).

Table 33. Example of narrative and quantitative rules from Northern Alabama BCG: BCG level 2 narrative and quantitative rules for macroinvertebrates and quantitative rules for fish in northern Alabama. Macroinvertebrate rules apply in all northern Alabama streams. Fish rules are applied by site class (PLA, RVP, and TV) and stream size (Small and Large).

Narrative Macroinvertebrate Rules for BCG Level 2

The sample is considered a level 2 condition if:
The number of all taxa in the sample is greater than 50–60 taxa and
The number of highly sensitive (attribute II) taxa is greater than 6–10 taxa and
The percentage of sensitive (attribute II+III) taxa is greater than 35%–40% of all taxa and
The number of sensitive (attribute II+III) EPT taxa is greater than 10–18 taxa and
The percentage of individuals in the 5 most abundant taxa is less than 60%–70% and
The percentage of individuals in the most abundant 5 tolerant (attribute IV, V, VI) taxa is less than 45%–55% OR the number of all taxa in the sample is greater than 70–80 taxa.
If any of these rules is not met at least half-way, the sample is level 3–6, depending on rules for those levels.

Macroinvertebrates: BCG Level 2	Quantitative Rule
# Total taxa	≥ 55 (50–60)
# Attribute II taxa	≥ 8 (6–10)
% Attribute II+III taxa	≥ 40% (35–45)
# Attribute II+III EPT taxa	≥ 14 (10–18)
% individuals in the most dominant 5 taxa	≤ 65% (60–70)
% individuals in the most dominant 5 tolerant taxa	≤ 50% (45–55) or Total Taxa > 75 (70–80)

Narrative Fish Rules for BCG Level 2

The sample is considered a level 2 condition if:
The number of all taxa in the sample is greater than 10–25 taxa in the PLA and RVP and
The number of highly sensitive (attribute I+II) taxa is greater than 0–4 taxa and
The number of sensitive (attribute I+II+III) taxa is greater than 5–10 in large TV sites and
The percentage of sensitive (attribute I+II+III) taxa is greater than 10%–25% and
The percentage of sensitive (attribute I+II+III) individuals is greater than 5%–30% and
The percentage of tolerant (attribute V+Va+VI) individuals is less than 15%–30% in the PLA and RVP and
The percentage of the most abundant Va or VI individuals is less than 30%–40% in the TV.
If any of these rules is not met at least half-way, the sample is level 3–6, depending on rules for those levels.

Fish: BCG Level 2	PLA		RVP		TV	
	Small	Large	Small	Large	Small	Large
# Total taxa	≥ 15 (10–20)	≥ 20 (15–25)	≥15 (10–20)	≥ 20 (15–25)	—	
# Attribute I+II taxa	> 2 (1–4)		> 0 (0–1)	> 2 (1–4)	> 1 (0–3)	≥ 2 (1–4)
# Attribute I+II+III taxa	—		—		—	> 7 (5–10)
% Attribute I+II+III taxa	≥ 20% (15–25)		≥ 15% (10–20)		≥ 20% (15–25)	
% Attribute I+II+III individuals	≥ 25% (20–30)		≥ 20% (15–25)		≥ 10% (5–15)	
% Attribute V+Va+VI individuals	≤ 25% (20–30)		≤ 20% (15–25)		—	
% Most dominant Attribute Va or VI individuals	—		—		≤ 35% (30–40)	

6.3.5 Application of the Biological Condition Gradient to Support Aquatic Life Use Assessments

Because biotic assemblages may respond to stressors differently depending on the mechanism of action, information from two or more assemblages provides more comprehensive insight into condition of the water, possible sources of stress, and potential for improvements. For example, the presence of small dams along streams and rivers alter natural flow and in stream habitat. These barriers prevent migration of some native species from rivers into streams. Likewise, presence of large reservoirs can introduce lake species into adjacent streams. Both of these impacts could result in a lower rating of biological condition using fish community data. An assessment of the benthic macroinvertebrate community of the same stream might result in a better biological condition rating if there are no additional stressors and physical habitat "as naturally occurs." This information would indicate that habitat and food source for fish exist and inform ADEM or other state agency decision makers that the stream may be a prime candidate for restocking of native species.

The BCG can be used by ADEM to characterize and communicate the biological conditions in the "least disturbed" reference reaches, aiding the interpretation of reference site quality relative to the absolute definitions of the BCG levels. "Least disturbed" reference sites are the best observable landscape and stream sites within a region. They can differ across regions of Alabama because development can be ubiquitous across entire regions of the state. The BCG can be used to interpret biological conditions in the "least disturbed" reference sites based on expert consensus in a manner that is transparent as long as expert judgment and the resulting decision rules are documented. For example, 57% and 44% of sites from ADEM's reference data set for two macroinvertebrate site classes, the Piedmont, Ridge, and Valley and the Southwest Appalachian regions, were assigned as BCG level 2 based on the benthic macroinvertebrate decision rules with the remainder of the sites primarily assigned as BCG level 3 (Figure 49). In contrast, only 13% of reference sites in the Interior Plateau were modeled as BCG level 2 and the majority of sites were assigned to BCG level 3. The differences in BCG levels among the reference sites of the three site classes illustrates how the "least disturbed" reference condition can have different biological meaning. BCG level 2 conditions support an aquatic community comparable to what would be expected under naturally occurring conditions with no or minimal anthropogenic impacts. The biological community characteristic of BCG level 3 includes loss of some native taxa and shifts in relative abundance of taxa relative to BCG level 2. Integration of the reference information and the BCG scale can be used to more clearly communicate to the public the quality of the reference condition for each region. In addition, existing indicators could be calibrated to the BCG scale to refine attainment thresholds. Despite the differences in reference site quality within the ADEM reference data set, there is a comparable relationship with the WDG in all three regions (Figure 50). The scatter observed with increasing WDG could, in part, be attributed to confounding effects and different mechanisms of action of multiple stressors as well as mitigating influence of BMPs that have been implemented.

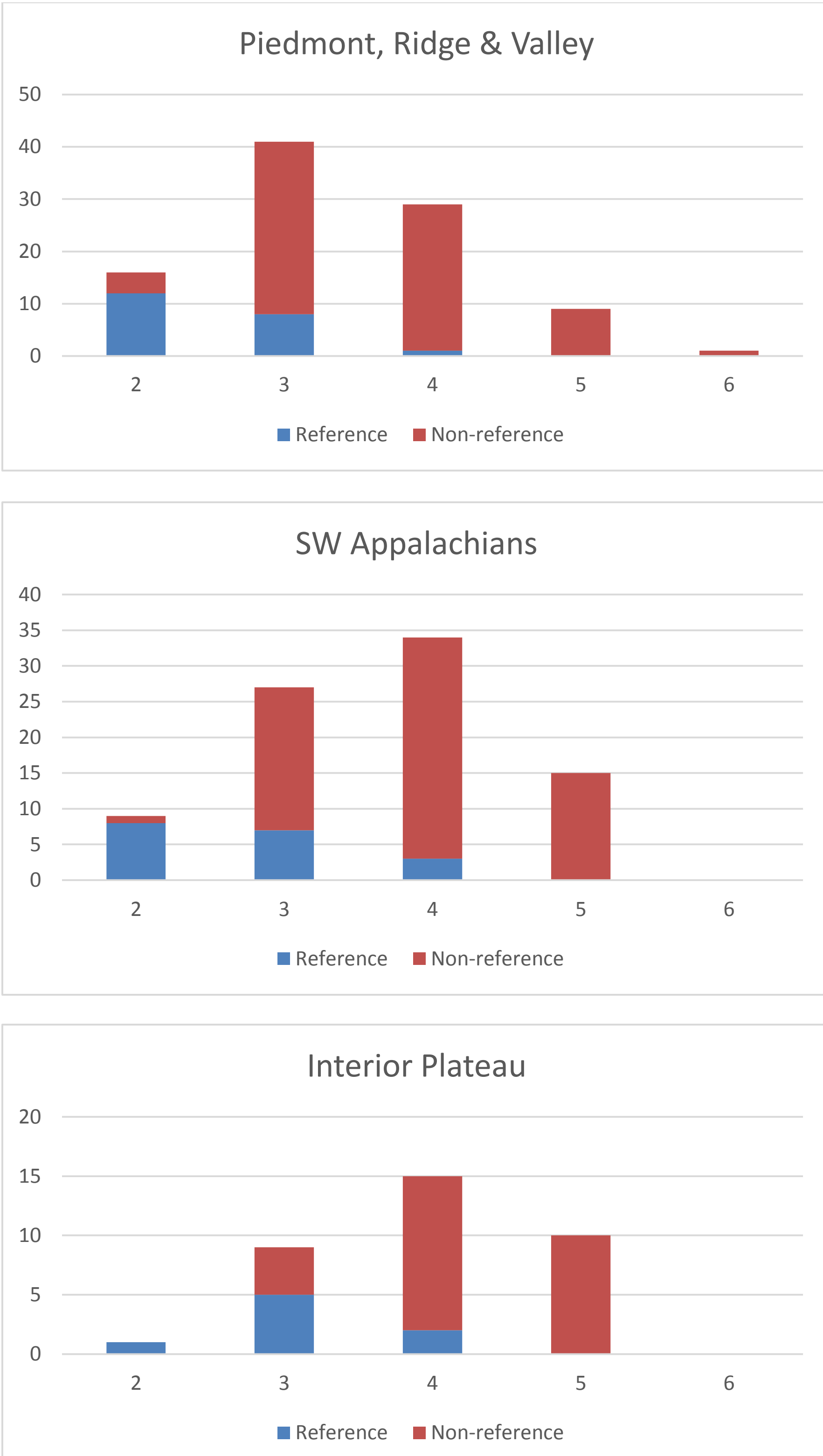


Figure 49. Frequencies of sites (y-axis) in each BCG level (x-axis) in each northern Alabama site class, showing reference sites as the blue portions of the bars. Distributions are based on sites monitored in ADEM's biological assessment program.

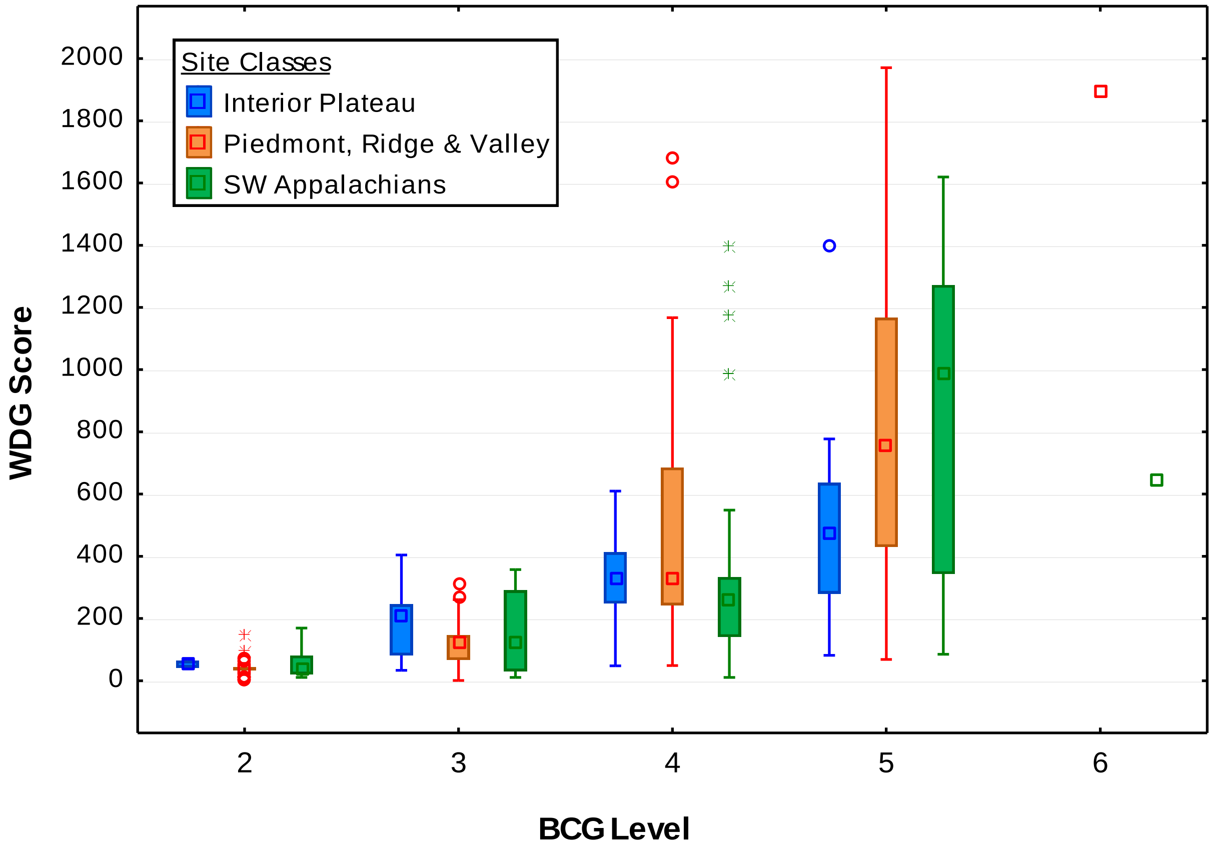


Figure 50. BCG scores and corresponding WDG scores for Northern Alabama. Distributions are based on sites monitored in ADEM's biological assessment program.

6.3.6 Future Applications

With the BCG model now available to characterize multiple levels of biological conditions, goals for protection of high quality waters and for improvements in degraded waters can be better defined. Currently, monitoring, assessment, and restoration focus on the most degraded watersheds throughout Alabama, leaving fewer resources to prevent threatened waters from degrading and becoming listed as impaired. Additionally, because success has typically been defined as a single threshold (i.e., attaining/nonattaining), incremental improvements in water quality and watershed conditions are not effectively measured and documented. Information that conditions are incrementally improving is valuable feedback to management, and stakeholders, including the public. Incremental changes can be observed with a shift in BCG levels or in index values associated with the BCG levels (Figure 51 and Figure 52).

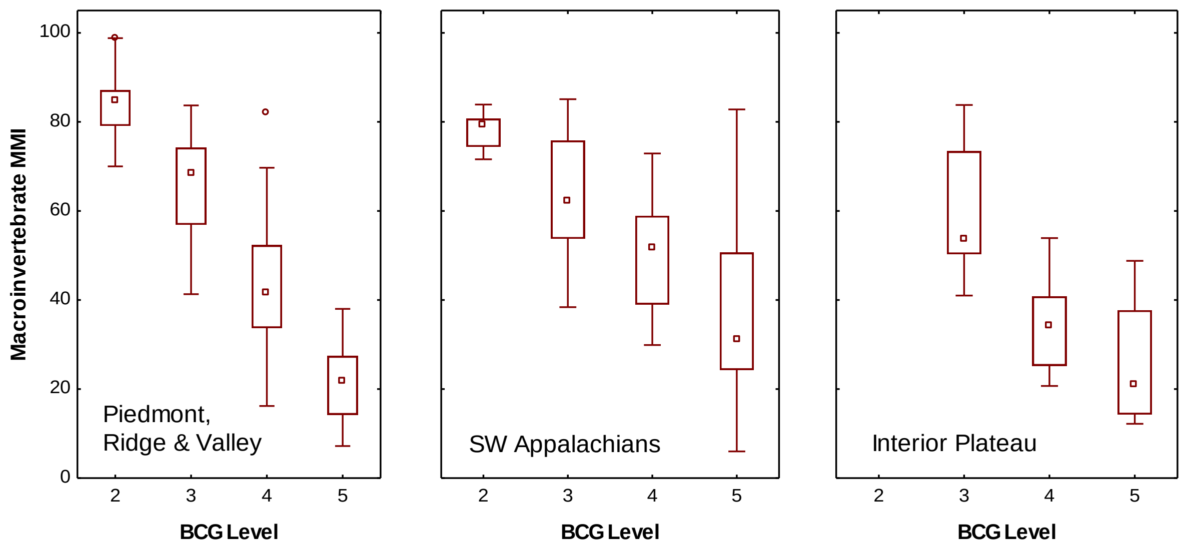


Figure 51. Alabama macroinvertebrate MMI distributions in site classes and BCG levels.

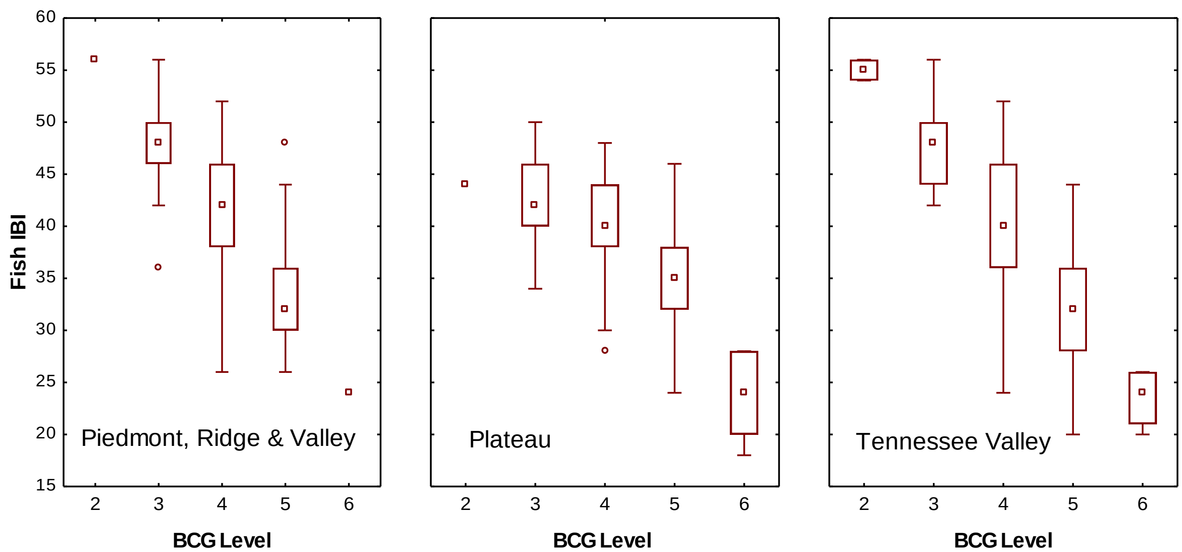


Figure 52. Alabama fish IBI distributions in site classes and BCG levels.

With the BCG, multiple condition levels can be recognized, and each can be associated with different resource status and management goals. For example, sites with BCG level 4, 5, or 6 conditions might be targeted for incremental improvements with interim milestones set based on next BCG level. Streams that score close to the next BCG level could be further prioritized for management actions. Such incremental improvements would document successful management strategies and actions and support adaptive management approaches. For sites supporting BCG level 2 conditions, the management goal might be protection so that the water body continues to support exceptional biological communities. BCG level 2 conditions could be identified using the predictive BCG models and/or the MMI and IBI scores.

As part of its Healthy Watersheds Program,[29] in 2011 EPA acknowledged the need to increase protection of U.S. waters and provided states with a framework and tools. In 2013, ADEM completed the Alabama and Mobile Bay Basin integrated assessment of watershed health (USEPA 2014b). The purpose of this project was to characterize the relative health of catchments across Alabama and the Mobile Bay Basin for the purpose of guiding future initiatives to protect healthy watersheds. The assessment synthesized disparate data sources and types to depict current landscape and aquatic ecosystem conditions throughout the Alabama/Mobile Bay Basin assessment area. The assessment included six distinct, but interrelated attributes of watersheds and the aquatic ecosystems within them, including landscape condition, habitat, hydrology, geomorphology, water quality, and biological condition. A total of 12 indicators were used to characterize the relative health of Alabama's watersheds. By integrating information on multiple ecological attributes at several spatial and temporal scales, it provided a systems perspective on watershed health. To compare the Healthy Watersheds Index (HWI) to BCG assessments, ADEM recalculated the HWI after removing the biological components from the calculation. The comparison showed a clear association between the non-biological HWI scores and the BCG scores (Figure 53). The ranges of HWI scores in each BCG level were similar among site classes, indicating that the BCG reflects differences in watershed integrity despite differences in landscape stressor intensity among site classes.

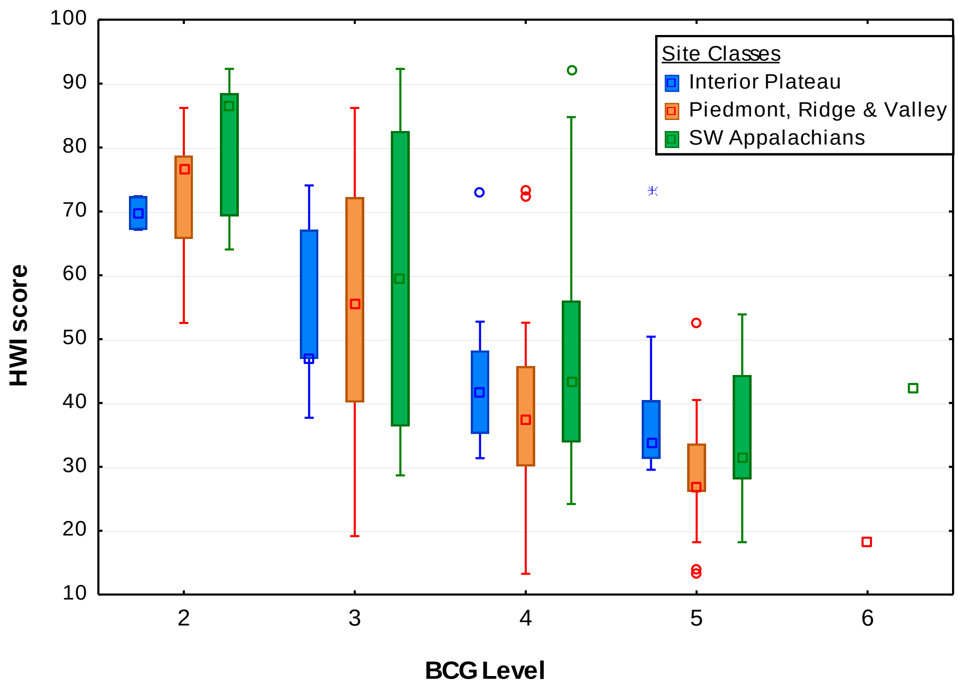


Figure 53. Distributions of Healthy Watershed Index (HWI) scores by macroinvertebrate BCG level and site class.

[29] More information on the Healthy Watersheds Program is available at: http://www.epa.gov/hwp. Accessed February 2016.

The most pervasive changes to watershed condition are predicted to come from population increase (changes in land and water use) and climate change (USEPA 2014b). Watershed vulnerability can be defined as a combination of an ecological system's exposure, sensitivity, and adaptive capacity to cope with changes in population and climate (IPCC 2007). The adaptive capacity of a watershed to cope with such changes is enhanced by connectivity of habitats and maintenance of floodplain, wetland, and other landscape features in their natural conditions to support natural hydrology and sediment supply. Vulnerability was characterized for Alabama watersheds using indicators of projected changes in precipitation, temperature, impervious cover, and water use (USEPA 2014b). Estimates of watershed health and vulnerability combined with the BCG level scores can potentially be used together to inform management decisions and priorities for protection and restoration.

6.3.7 Conclusion

ADEM developed a BCG model to expand the technical capability of its biological monitoring and assessment program, with four key results. First, ADEM has been able to use the BCG to more accurately characterize the quality of reference sites relative to natural conditions (e.g., no or minimal anthropogenic disturbance). Second, in conjunction with biological indices, ADEM has used the BCG as a tool to help identify high quality streams, evaluate recovery potential of degraded streams, propose incremental biological goals for improvements, and track improvements. Third, ADEM is better able to convey to the public and decision makers more detail about the aquatic community to assist both the public and water quality managers in prioritizing areas for protection and restoration.

Finally, ADEM has found that adding fish community assessments to its biological assessment program produces more robust and comprehensive assessments of aquatic life (USEPA 2013a). Fish assessments are the primary biological indicator used to assess the status of threatened and endangered aquatic species within the state. Macroinvertebrate and fish assessments are generally conducted at different sites to make the most of limited resources and enable ADEM and partner agencies to assess biological conditions at more sites throughout the state. The two assemblages are sensitive to different stressors because of differences in the life cycles and motility of fish and benthic macroinvertebrates. The potential for different kinds of stress, the presence of threatened and endangered species, watershed area, and depth are all factors used to determine which assemblage will be assessed at each site. The BCG provides a common interpretive framework for benthic and fish assemblage data so both sets of information could be mapped on a common assessment scale and the information used to inform management decisions.

6.4 Minnesota: More Precisely Defining Aquatic Life Uses and Developing Biological Criteria

6.4.1 Key Message

Most surface waters in Minnesota are protected for aquatic life and recreation to meet the objectives set forth in CWA section 101(a). In the state, there are two primary sub-classes of streams protected for aquatic life, including a cold water stream class (2A) and a warm water stream class (2B). While the current system of beneficial uses and WQS has served Minnesota well, advances in the fields of biological assessment have led to the recognition that among the diversity of water body types there are variable biological conditions. For example, within rivers and streams, factors such as water body size, geographic location, hydrology, water temperature, and stream gradient influence chemical, physical, and biological composition. The Minnesota Pollution Control Agency (MPCA) recognized that effective water quality management requires a more comprehensive approach in which goals for water quality protection are tailored to specific water body types and uses. In response to these challenges, MPCA is proposing to modify its beneficial use framework for aquatic life. The new framework will allow for better goal-setting processes through the application of a framework that recognizes tiers, or levels, of aquatic life-use based on a stream's type and potential. MCPA is using the BCG to describe existing biological conditions and help provide the technical basis for assigning streams to ALU classes.

6.4.2 Background

MPCA's collection of biological water quality information began in the 1960s as part of an effort to monitor the conditions of state waters and since that time the state has developed a robust biological assessment program (USEPA 2013a). Over the past two decades, MPCA has routinely monitored both fish and benthic macroinvertebrates in streams, and, in combination with assessment of chemical and physical parameters, has used this information to assess the integrity of streams (MPCA 2014b). In the mid-1990s MPCA developed IBIs for fish (F-IBI) and benthic macroinvertebrates (M-IBI) to characterize the health of biological communities, identify stressors, select management actions to protect and restore water bodies, and determine how effective management actions are in meeting those goals. The initial IBIs developed were supported by narrative statements in the state's regulatory language that identified how to calculate an IBI. In 2003 and 2004, IBIs were developed for streams in specific basins of the state, and subsequently MPCA developed IBIs that could be applied statewide (MPCA 2014c, 2014d). Both the M-IBI and F-IBI used today are calibrated for a number of stream environments (e.g., large rivers, moderate-sized streams, headwaters, low-gradient streams, and cold water streams) (MPCA 2014c, 2014d). The IBIs for different stream types minimize the effects of natural differences between streams in order to enhance the signal from anthropogenic stressors. For example, the St. Louis River, a large river in northern Minnesota, naturally has a very different fish fauna compared to a small cold water stream in southern Minnesota such as Beaver Creek (Figure 54). Because the fish communities are naturally different in these habitats, IBI models need to be specific to the stream type so that appropriate expectations for healthy communities can be established. Since 2007, MPCA has monitored the state's rivers and lakes using a 10-year rotating watershed approach.

Minnesota's WQS classify state waters according to their designated beneficial uses (e.g., aquatic life, recreation, drinking water), and the state applies chemical, physical, and biological criteria to protect designated uses. Currently, the majority of surface waters in Minnesota are classified as Class 2,

Figure 54. Left: St. Louis River; Right: Beaver Creek.

protection of aquatic life and recreation[30] (i.e., the "General Use" goal). For streams and rivers, class 2 waters are further distinguished as Class 2A (aquatic life cold water habitat) or Class 2B (aquatic life warm water habitat). Despite the application of chemical, physical, and biological criteria, state scientists determined that a single biological threshold does not reflect existing conditions in high quality waters, nor set attainable restoration goals for degraded waters. For example, the West Branch of the Little Knife River (Figure 55) in the Lake Superior drainage in Minnesota supports fish and macroinvertebrate assemblages that would be expected in environments comparable to BCG level 1 or 2. A contrasting example is Judicial Ditch 7 in southeastern Minnesota (Figure 55). Fish and macroinvertebrate assemblages in this stream do not meet the stream's current aquatic life goal, which is estimated to be comparable to BCG level 4, because it is maintained for drainage. The activities associated with maintaining this ditch for drainage remove the habitat necessary to support natural aquatic assemblages and might limit attainment of the designated ALU. A use attainment analysis (UAA) will support determination of the highest attainable use for these types of streams, and the BCG could provide the basis for setting incremental restoration targets and tracking improvements.

Figure 55. Left: West Branch Little Knife River; Right: Judicial Ditch 7.

[30] A full definition of *Class 2 water* can be found in Minnesota Administrative Rule 7050.0140, Subp. 3. https://www.revisor.leg.state.mn.us/rules/?id=7050.0140. Accessed February 2016.

6.4.3 Tiered Aquatic Life Uses and Biological Criteria Development

Over the past ten years, state scientists have sought an approach that would capitalize on the state's wealth of biological monitoring data and more specifically define the ALUs of rivers and streams in Minnesota. MPCA is revising the state WQS to more accurately designate ALUs and establish multiple levels (or goals) for aquatic life conditions in the WQS (in Minnesota this is known as the tiered aquatic life use (TALU) framework). Using this framework, Minnesota is proposing to classify rivers and streams based on the best attainable biological condition for a water body. The state is also proposing to subcategorize its designated ALU categories to best reflect a stream or river's current conditions and its ecological potential. This approach requires knowledge of the current condition of water bodies and the stressors affecting them (MPCA 2012). In order to develop TALUs and associated biological criteria, MPCA has capitalized on a variety of past work, including stream classification, IBI development, an HDS, and the BCG (MPCA 2014b). The BCG was used to interpret current conditions and set expectations for biological communities across the state. IBIs are used to determine the biological conditions of state rivers and streams and to determine which ALU best describes the highest attainable biological conditions in a specific water body.

MPCA's application of TALUs will subdivide Class 2 streams into three designated use class tiers (MPCA 2014e):

- Exceptional uses—"High quality waters with fish and invertebrate communities at or near undisturbed conditions."
- General uses—"Waters with good fish and invertebrate communities that meet minimum restoration goals."
- Modified uses—"Waters with legally altered habitat that prevents fish and invertebrate communities from meeting minimum goals."

For each designated use class tier, MPCA has developed biological criteria using biological, chemical, physical, and land use data collected during the 1995–2010 period. MPCA used a multiple lines of evidence approach that included use of the BCG and the reference condition.

In order to identify reference streams, MPCA first calculated an HDS, an index that measures the degree of human activity upstream of and within a stream. MPCA defined stream reference sites as those with an HDS score of 61 or greater; this is a defined least disturbed condition (the upper 25% of the HDS distribution). The reference streams are least influenced by stressors within the context of the current landscape condition of Minnesota (Stoddard et al. 2006), as far as practical from urban areas, point sources, feedlots, and other sources. MPCA also identified a subset of reference streams that satisfied "minimally disturbed" in the northern part of the state where widespread and long-term human disturbance is much less than in the south. MPCA compared the IBI scores for reference and non-reference sites. While MPCA identified some concerns with applicability of the reference condition approach in southern Minnesota due to widespread, high levels of land use and development, the agency determined that reference data sets were sufficient to develop biological criteria in the northern regions and in cold water classes (MPCA 2014b). Reference conditions for the southern region might require an alternate approach to more precisely characterize least disturbed conditions.

During 2009–2012, expert panels were assembled to develop BCG models for both macroinvertebrate and fish assemblages (Gerritsen et al. 2013). The conceptual BCG model (Davies and Jackson 2006) was calibrated by these expert panels using regional data for each of the two assemblages. The narrative descriptions for the different BCG condition levels were used by MPCA to describe each of the three designated use class tiers proposed in the revision to its WQS regulation:[31]

- Exceptional Use—"Evident changes in structure due to loss of some rare native taxa; shifts in relative abundance; ecosystem level functions fully maintained."
- General Use—"Overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes."
- Modified Use—"Sensitive taxa markedly diminished; conspicuously unbalanced distribution of major taxonomic groups; ecosystem function shows reduced complexity & redundancy."

The MPCA expert panels characterized and calibrated the BCG for both benthic macroinvertebrates and fish for seven classes of warm water streams and two classes of cold and coolwater streams. A summary of the narrative rules includes:

- Taxa richness declined from BCG level 1 to level 6. All level 1 sites were large water bodies (rivers), and might be more influenced by size than by condition
- Attribute I taxa were characteristic of BCG level 1, occurred occasionally in BCG level 2, and were generally absent in levels 3–6
- All sensitive taxa (attributes I, II, and III combined) are common and abundant in levels 1 and 2, somewhat reduced in level 3, decline markedly in level 4, and have almost disappeared from levels 5 and 6.
- Intermediate taxa (attribute IV) are nearly constant throughout the gradient, but are reduced in level 6.
- MPCA divided the tolerant fish category into two: tolerant taxa (attribute V), and highly tolerant taxa (attribute V-a), as well as highly tolerant nonnative (attribute VI-a). The highly tolerant subgroups increased in abundance, dominance and variability at BCG levels 4 to 6, although the natives are represented at all levels.

An example of quantitative BCG rules derived for fish in the two river classes is shown in Table 34.

[31] Information about Minnesota's WQS process is available at: http://www.pca.state.mn.us/index.php/water/water-permits-and-rules/water-quality-standards.html. Accessed February 2016.

Table 34. Decision rules for fish assemblages in two classes of Minnesota rivers. Rules show the ranges of fuzzy membership functions. N indicates the number of sites for a given BCG level and stream class in the calibration data set.

Metric	Prairie Rivers	Northern Forest Rivers	
BCG Level 1	N=2	N=3	
Total taxa	> 25–35	> 16–24	
Endemic taxa (Att I)	Present	Present	
Att I+II taxa	> 2–5	> 1–2	
Att I+II+III % taxa	> 45%–55%	> 35%–45%	
Att I+II+III % ind	> 25%–35%	> 45%–55%	
Att Va or VIa Dominance		< 7%–13%	
Tolerant % ind (V + Va + VIa)	< 3%–7%		
Highly tol % ind (Va + VIa)		< 7%–13%	
BCG Level 2	N=6	N=15	
Total taxa	> 16–24	> 6–10	
Att I+II taxa	Present	-	
Att I+II+III % taxa	> 35%–45%	> 25%–35%	
Att I+II+III % Ind	> 15%–25%	> 25%–35%	
Att Va or VIa Dominance		< 7%–13%	
Highly tol % ind (Va + VIa)	< 7%–13%	< 7%–13%	
BCG Level 3	N=25	N=11	
Total taxa	> 11–16	> 6–10	
Att I+II+III % taxa	> 15%–25%	> 15%–25%	
Att I+II+III % Ind	> 7%–13%	> 7%–13%	
Tol % ind (V + Va + VIa)	-	< 25%–35%	
Att Va or VIa Dominance	< 7%–13%	< 10%–20%	
Highly tol % ind (Va + VIa)	< 25%–35%	-	
BCG Level 4	N=31	N=16	
		Alt 1	**Alt 2**
Total taxa	> 11–16	> 6–10	= alt 1[1]
Att I+II+III % taxa	10%–20%	> 15%–25%	> 7%–13%
Att I+ II+III % Ind	0%–1%	> 3%–7%	present
I+II+III+IV % Ind			
Att Va or VIa Dominance	< 35%–45%	< 25%–35%	= alt 1[1]
Tol % ind (V + Va + VIa)		n/a	< 30%–40%
Highly Tol % ind (Va + VIa)	< 45%–55%	< 35%–45%	= alt 1[1]
BCG Level 5	N=12	N=2	
Total taxa	> 11–16	6–10	
Att I+II+III+4 % Taxa			
Att Va or VIa Dominance	< 65%–75%	< 35%–45%	
Highly tol % ind (Va + VIa)		< 55%–65%	
BCG Level 6 (no rules)	N=1	N=0	

[1] "= alt 1" the rule is the same as given under Alt 1 for this metric

MPCA then calibrated the BCG with the state's index for biological assessment of Minnesota's warm water and cold water streams for both the fish and macroinvertebrate assemblages (Figure 56). MPCA has used this information to develop draft numeric biological criteria that would be applied to each designated use class tier—thus directly linking the ALU goal with the state's assessment method (Figure 57). In December 2015, MPCA held a formal public comment period on a proposed revision to the state WQS that would include TALUs.

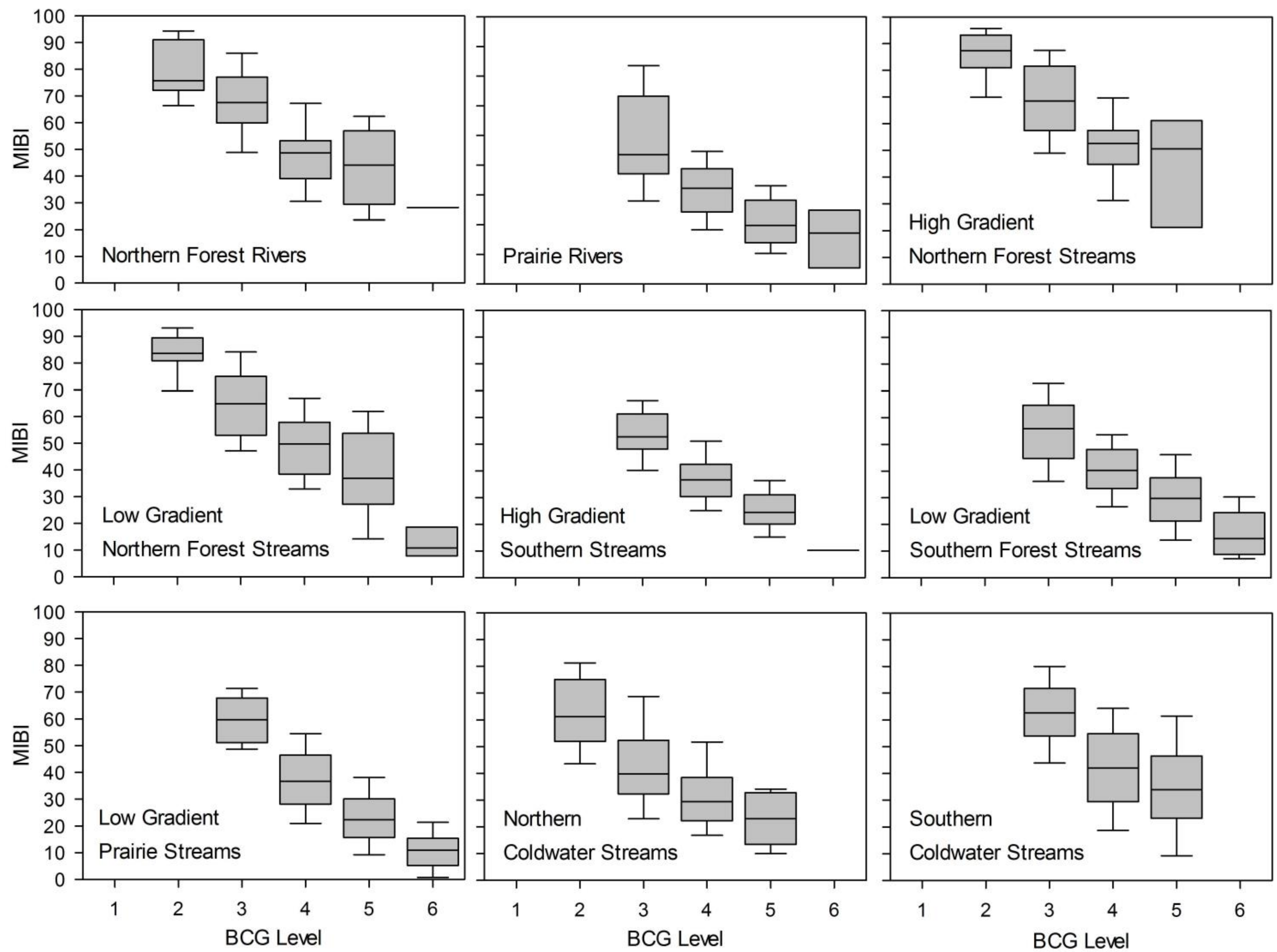


Figure 56. Frequency distributions of IBI scores by BCG level for macroinvertebrate stream types using data from natural channel streams sampled 1996–2011. Symbols: upper and lower bounds of box = 75th and 25th percentiles, middle bar in box = 50th percentile, upper and lower whisker caps = 90th and 10th percentiles. MPCA also did a calibration of fish index scores with BCG levels assigned to sites.

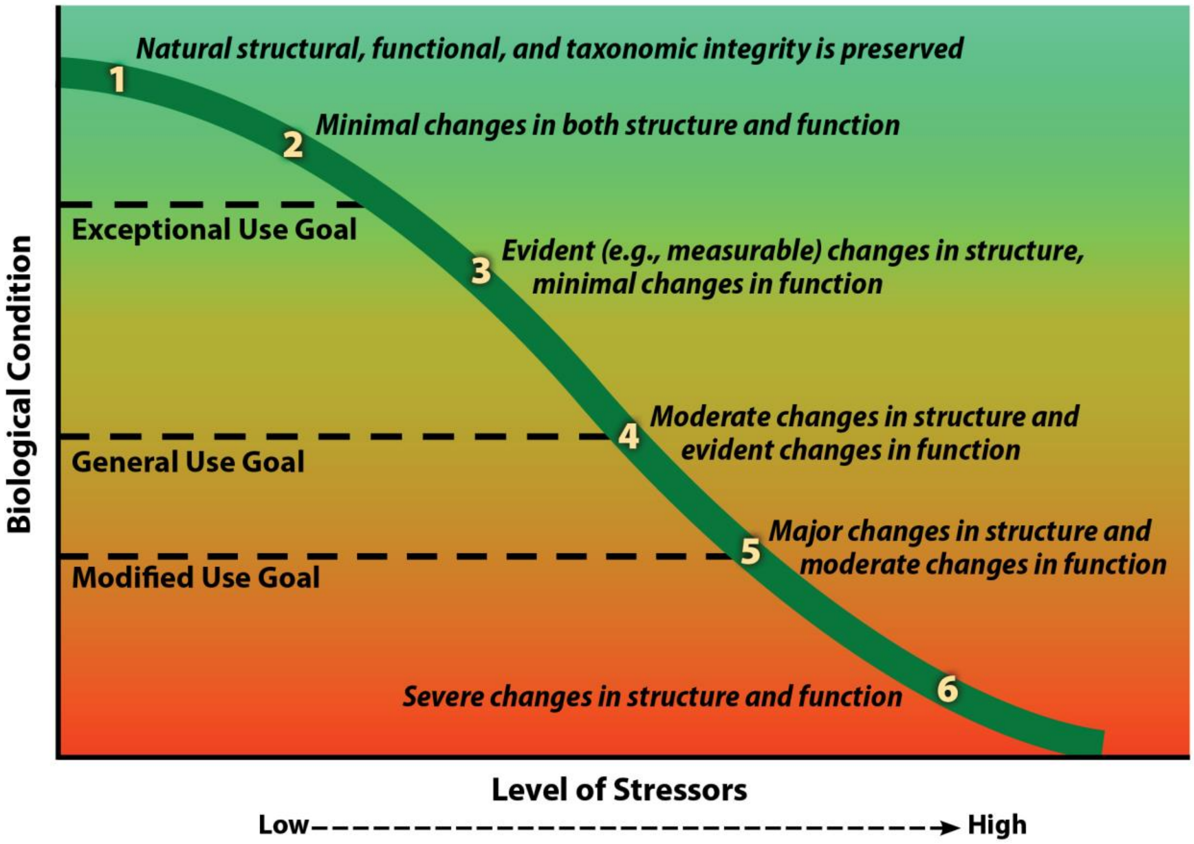


Figure 57. BCG illustrating the location of proposed biological criteria (black dotted line) for protection of Minnesota's TALU goals (Exceptional, General, Modified) (Source: MPCA 2014b).

6.4.4 Benefits of the Biological Condition Gradient

Because the BCG provides a common framework to interpret changes in biological condition regardless of geography or water resource type, Minnesota will be able to make more accurate determinations and classifications of its aquatic resources on a statewide basis. The state will be in position to make decisions on aquatic life designations based on robust and detailed ecological data and information. Another advantage of the BCG is that it provides a means to communicate with the public about existing conditions and potential for improvement for specific water bodies. BCGs were developed for each of Minnesota's aquatic resource classes for streams (e.g., cold water and warm water streams). The development of warm water BCG models involved input from biological experts familiar with biological communities in Minnesota from the MPCA and Minnesota Department of Natural Resources. BCG models were developed for fish and macroinvertebrates for each of the seven warm water stream classes. A cold water BCG involved experts from Minnesota, Wisconsin, Michigan, and several tribes located in those states. In Minnesota this effort included two classes each for fish and macroinvertebrates. Model development for each class involved reviewing biological community data from monitoring sites and then assigning that community to a BCG level. A sufficient number of samples were assessed to develop a model that can duplicate the panel's BCG level assignments. Using the BCG and reference conditions permits MPCA to provide more detailed descriptions of the expected biota for each ALU and to develop biological criteria that are protective, consistent, and attainable across the state (MPCA 2012). These accomplishments will help Minnesota achieve several key goals described below.

Refinement of Biological Standards

Numeric water quality criteria that are codified in the Minnesota WQS are currently based on chemical and physical criteria such as DO, temperature, and pH. These criteria do not directly measure the condition of biological communities that include fish, insects, mussels, aquatic plants, and algae. Biological communities can be monitored as a direct measure of the response of the biota to a wide range of physical and chemical stressors and provide a quantitative measure of the cumulative and synergistic impacts of multiple stressors over time. A major goal of Minnesota's water quality management program is to protect the fish, invertebrates, and other aquatic organisms in the state's waters. Therefore, it is sensible that a direct measurement of these communities is used to monitor their condition.

Ability to Address Natural Variation

One of the strengths of Minnesota's approach is the ability to address the natural variation in water resources across the state. Minnesota's diverse water resources mean that refined biological monitoring tools are needed to reduce errors in assessment and management. For example, streams along the shore of Lake Superior in northern Minnesota are very different from streams in southern Minnesota such that, under natural conditions, the biological communities in streams in each location are expected to be different. The Minnesota BCG framework takes into account these natural differences and requires that comparisons be made between streams with naturally similar biological communities. As the state's database is built through long term monitoring, Minnesota will be able to define current, or baseline, conditions and be in a better position to discern shifts in species composition and structure due to climate change impacts.

Identification of Reference Condition Quality

The biological monitoring program in Minnesota relies on BCG models and the reference condition approach to set expectations for water bodies. The BCG provides a common "yardstick" of biological condition that is rooted in the natural condition. As a result, the BCG can be used to develop biological criteria that are consistent across regions and stream types in Minnesota—particularly important for a state where the range of existing quality is regionally distinct and extreme (i.e., undisturbed to highly disturbed conditions). The reference condition approach identifies water bodies that are least disturbed and uses them to establish the reference condition. Once this reference condition has been established, water bodies with unknown condition can be compared to this baseline. If the condition of the water body is lower than that of the reference condition, it would be considered impacted or stressed. The use of a reference condition relies on the development of accurate expectations for least disturbed sites. The BCG provides a framework for assessing the quality of reference sites relative to undisturbed conditions and can be used to interpret the quality of reference sites, including reference sites in regions where the least disturbed conditions include sites with moderate to higher levels of stress. In these regions, such as in southern Minnesota, the BCG was used to help develop protective ALU goals (MPCA 2014b, 2014e).

Protection of High Quality Water Resources

Minnesota's classification framework and BCGs will be applied in conjunction with another element of states' antidegradation policy. This policy requires:

- Maintenance of existing uses;
- Prevention of degradation of water quality that exceeds levels necessary to support the protection and propagation of aquatic life and recreation unless the state finds that lowering of

water quality is necessary to accommodate important economic or social development (Tier 2 protection); and

- Maintenance of water quality needed to protect outstanding resource waters (Tier 3 protection).

Minnesota is planning to propose a higher tier of ALU (i.e., exceptional use goal) to protect high quality biological communities. Once it has been established that a water body is meeting the requirements associated with an exceptional water resource, the resource needs to be protected to maintain that status. The BCG provides a framework with which to identify candidate high quality streams and rivers for designation as exceptional resources.

Setting Expectations for Modified Water Resources

There are water resources in Minnesota that will not in the near future meet the CWA interim goals due to historical or legacy impacts. These legacy impacts include streams under drainage maintenance or other irreversible hydromodification that preclude attainment of water body goals (e.g., channelized streams and ditches). The BCG provides a framework to monitor and help set realistic expectations for waters that are unlikely to meet ALU goals due to legacy impacts and have been designated as modified water resources. Additionally, as conditions improve, the BCG provides a framework to document and acknowledge these improvements to reflect existing conditions.

6.4.5 Conclusion

In conjunction with numeric biological indices developed for macroinvertebrates, the BCG allows Minnesota to set consistent and protective ALU goals and numeric biological criteria across the state despite the heterogeneity of its water bodies. This heterogeneity is due both to natural conditions and human disturbance, and the BCG provides a framework to characterize and communicate these differences. The BCG described in this case study is applicable to streams and wadeable rivers. Minnesota is currently developing a BCG and biological criteria for lakes using fish assemblage information.

6.5 Maine: Development of Condition Classes and Biological Criteria to Support Water Quality Management Decision Making

6.5.1 Key Message

Clear, technically rigorous goal statements have provided Maine with an effective framework to improve biological condition of streams and rivers. Maine has established four ALU classes (Classes AA/A/B/C) with different ecological expectations. The classes span the range from Maine's interpretation of the CWA interim goal to the ultimate CWA objective "to restore and maintain chemical, physical and biological integrity" (Class AA/A). All rivers and streams in Maine are assigned to one of the four classes in Maine's WQS for planning and management purposes. These TALUs and numeric biological criteria have enabled Maine to inject critical biological information into all aspects of water quality management. Along with the practical experience and scientific advancements demonstrated by other states with strong biological assessment programs, Maine's approach to classification and biological criteria development provided the template for the conceptual BCG (Davies and Jackson 2006). In turn, Maine continues to strengthen and develop its biological assessment program to address other water bodies and include measures of the algal communities in its assessments. The BCG is being incorporated as part of its "toolbox" to accomplish these tasks.

6.5.2 Background

Since the 1960s, prior to adoption of the CWA, Maine water quality law has had a tiered structure based on observations of gradients of water quality conditions. In 1986, Maine revised its water classification law and added TALUs to maintain and restore the structure, function, and biological integrity of aquatic life communities. Maine's TALUs were based on concepts of John Cairns, H.T. Odum, and others who observed declines in biological condition in response to gradients of increasing stressors (Ballentine and Guarraia 1977; Odum et al. 1979, Cairns et al. 1993; Karr and Chu 2000). The four narrative TALU standards in Maine's water classification law describe conditions across a biological gradient ranging from "as naturally occurs" (Classes AA and A) to "maintenance of structure and function" (Class C). Class C is the lowest ALU designation allowed in the state and consistent with Maine's interpretation of the CWA fishable/swimmable interim goal (Table 35; M.R.S.A Title 38 Article 4-A § 464-466). Maine's TALUs for fresh surface waters apply to streams, rivers, and wetlands. Maine has similar TALUs for coastal marine waters (SA, SB, SC). Maine has established a single class for lakes that is equivalent to Class A. Maine's TALUs are based on tiers of biological condition along observed human disturbance gradients. Such stressor-response relationships are also the foundation of the later development of the BCG.

Maine's TALUs are supported by ecologically-based definitions in the law. The narrative definitions in Maine law establish the biological characteristics that are required to attain the standards of each class (Table 35). Class AA and Class A have the same "*as naturally occurs*" aquatic life goals and will hereafter be referred to as Class AA/A; Class AA is more restrictive in allowable permitted activities. For example, no dams or discharges are allowed in Class AA waters. Maine's assessed streams and rivers are predominantly classified as either Class AA/A or B waters, 48.6% and 51%, respectively. Class A/AA waters have been interpreted by Maine as comparable to BCG levels 1 and 2 and class B waters are equivalent to BCG level 3. Less than 1% of Maine's streams and rivers are classified as Class C waters, which have been deemed as comparable to BCG level 4. These waters are primarily in urbanizing areas or downstream of significant point sources. Figure 58 summarizes relationships between Maine's narrative biological, chemical, and physical standards and shows Maine's TALUs in relation to the BCG.

Table 35. Criteria for Maine river and stream classifications and relationship to antidegradation policy

Class	DO criteria	Bacteria criteria	Habitat narrative criteria	Aquatic life narrative criteria*** and management limitations/restrictions	2012 Percentage of Maine waters designated in class ****	Corresponding federal antidegradation policy tiers
AA	As naturally occurs	As naturally occurs	Free-flowing and natural	As naturally occurs**; no direct discharge of pollutants; no dams or other flow obstructions.	3.6%	3 (Outstanding National Resource Water [ONRW])
A	7 ppm; 75% saturation	As naturally occurs	Natural**	Discharges permitted only if the discharged effluent is of equal to or better quality than the existing quality of the receiving water; before issuing a discharge permit the Department shall require the applicant to objectively demonstrate to the department's satisfaction that the discharge is necessary and that there are no reasonable alternatives available. Discharges into waters of this class licensed before 1/1/1986 are allowed to continue only until practical alternatives exist.	45%	2 ½
B	7 ppm; 75% saturation	64/100 mg (g.m.) or 236/100 ml (inst.)*	Unimpaired**	Discharges shall not cause adverse impact to aquatic life** in that the receiving waters shall be of sufficient quality to support all aquatic species indigenous** to the receiving water without detrimental changes to the resident biological community.**	51%	2 to 2 ½
C	5 ppm; 60% saturation; and 6.5 ppm (monthly avg.) when temperature is ≤ 24 °C	125/100 mg (g.m.) or 236/100 (inst.)*	Habitat for fish and other aquatic life	Discharges may cause some changes to aquatic life**, provided that the receiving waters shall be of sufficient quality to support all species of fish indigenous** to the receiving waters and maintain the structure** and function** of the resident biological community. **	0.4%	1 to 2

Source: Maine DEP (modified). http://www.maine.gov/dep/water/monitoring/classification/index.html. Accessed February 2016.
Notes:
* g.m. = geometric mean; inst. = instantaneous level.
** Terms are defined by statute (Maine Revised Statutes Title 38, §466).
*** Numeric biological criteria in Maine regulation Chapter 579, Classification Attainment Evaluation Using Biological Criteria for Rivers and Streams http://www.maine.gov/dep/water/rules/index.html. Accessed February 2016.
**** Source: 2012 Maine Integrated Water Quality Report, http://www.maine.gov/dep/water/monitoring/305b/2012/report-final.pdf. Accessed February 2016.

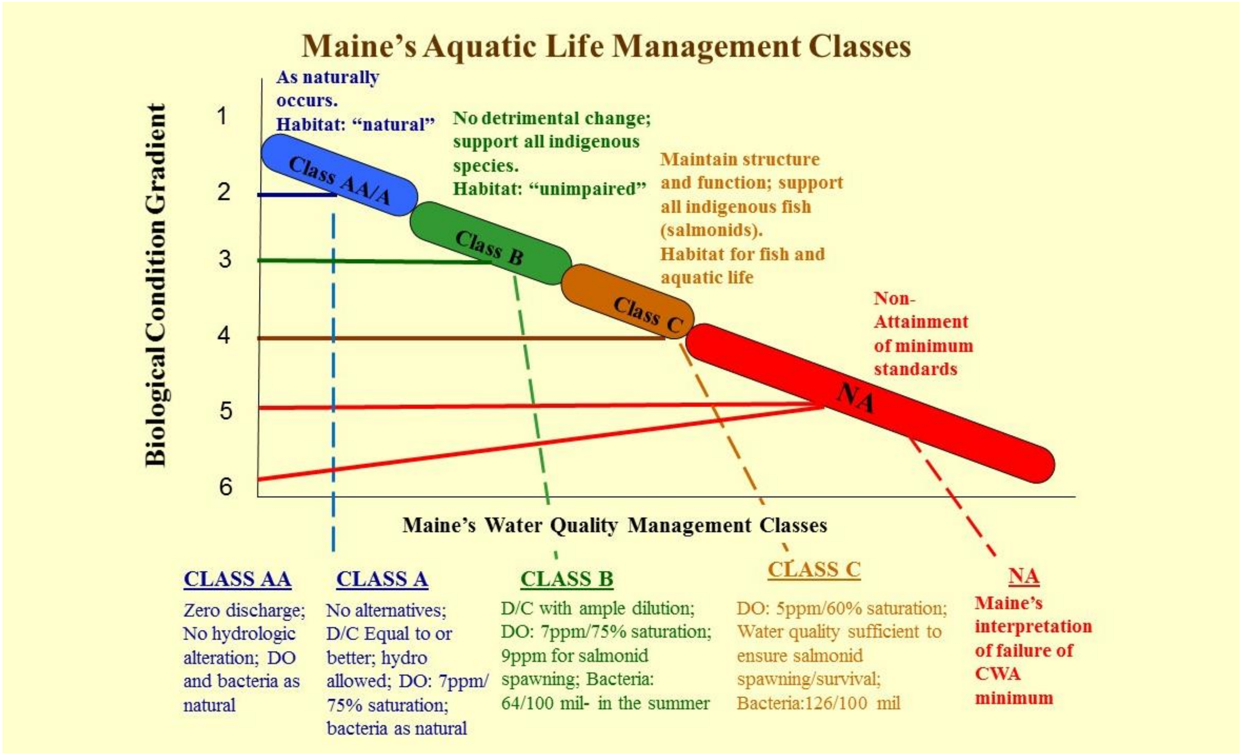


Figure 58. Relation between Maine TALUs, the BCG, and Maine's other water quality standards and criteria. Class AA/A is approximately equivalent to BCG levels 1 and 2. Classes B and C approximate BCG levels 3 and 4, respectively. Non-attainment conditions below Class C are approximately equivalent to BCG levels 5 and 6.

6.5.3 Maine's Numeric Biological Criteria and Tiered Aquatic Life Uses

In 2003, Maine adopted numeric biological criteria in rule for rivers and streams, based on assessment of benthic macroinvertebrates (State of Maine 2003; Shelton and Blocksom 2004; Davies et al. In press). Technical details describing development of the statistical biological criteria models are found in Chapter 4 of this document and in Davies et al. (In press). In short, MEDEP utilized expert consensus to establish four *a priori* groups corresponding to Maine's TALUs, and developed and tested a linear discriminant model (LDM) to predict the probability of a sample attaining ALU goal conditions (Class AA/A, Class B, and Class C). The fourth group, termed "non-attainment" (NA) represents samples that are in poorer condition than Class C. The LDM and accompanying provisions for application are codified in rule and constitute Maine's numeric biological criteria.[32] When confirmed (e.g., by re-sampling and review of data results) that a stream reach fails to attain its assigned water quality goal, the water body segment is listed as impaired on the state's 303(d) list (Table 36). State law requires that water bodies be considered for upgrade to a higher class if they are found to be consistently attaining the standards of that higher classification.

[32] http://www.maine.gov/dep/water/rules/index.html. Accessed February 2016.

Table 36. Examples of how numeric biological criteria results determine whether or not a water body attains designated ALUs in Maine

Legislative Class	Monitoring Result	Attains Class?	Next Step
A	A	Yes	--
C	B	Yes	Review for upgrade
A	B	No	303(d) list as impaired if confirmed
B	NA	No	303(d) list as impaired if confirmed

MEDEP also conducts biological assessments of stream algal, wetland macroinvertebrate, and wetland phytoplankton and epiphytic algal assemblages (Danielson et al. 2011, 2012). MDEP used Maine's narrative biological criteria and the BCG as the foundation of biological assessment models for stream algae and wetland macroinvertebrates. A first step in model-building was to empirically compute tolerance values for algal and macroinvertebrate species that had been collected in Maine's monitoring program. After computing tolerance values, the species were grouped into the BCG framework's sensitive, intermediate, and tolerant attribute groups. MEDEP then modified the BCG framework for stream macroinvertebrates for stream algae and wetland macroinvertebrates, describing how those assemblages empirically respond to anthropogenic stressor gradients. MEDEP used the BCG and tolerance metrics along with the narrative biological criteria and other metrics to build predictive biological assessment models for the additional assemblages. MEDEP has completed LDM statistical models to predict TALU attainment for both stream algal and wetland macroinvertebrate community data. These models currently are used to help interpret narrative biological criteria. Following adequate testing and standard public review protocols, MEDEP intends to amend the Maine Biological Criteria Rule[33] to include the stream algal and wetland macroinvertebrate models as numeric biological criteria.

In summary, numeric biological assessment models, when codified in the MEDEP biological criteria rule (as for stream macroinvertebrates), or when used as an objective corroboration of expert judgment (as for stream algae and wetlands), provide a transparent and standardized quantitative means for determining attainment of TALUs in Maine WQS. Numeric biological criteria have enabled Maine to use biological information to support multiple water quality management information needs and decision making. Examples of applications follow.

6.5.4 Goal-based Management Planning to Optimize Aquatic Life Conditions

As described in section 6.5.2, the Maine State Legislature revised Maine's WQS and classification law in 1986 (M.R.S.A Title 38 Article 4-A § 464-466) establishing narrative biological criteria for four ALU classes for rivers and streams. This law set in motion a process involving the public, the state environmental agency, and the Maine legislature to assign all Maine waters to an appropriate goal classification. All available monitoring data and information about then-current biological and/or water quality conditions were used to initially propose the statutory classes for stream and river segments for the 1986 law. Many waters that lacked current monitoring data retained their previous water quality goals (generally Class B, except for some urban or industrialized areas, which were Class C) until monitoring data or other evidence was found to recommend a different (and in most cases higher) class.

[33] See Code of Maine Rules, MEDEP, Chapter 579, http://www.maine.gov/dep/water/rules/index.html. Accessed February 2016.

Maps spanning the period between 1987 (Figure 59) and 2012 (Figure 60) show the past and present distribution of water quality classifications. Approximately 99% of Maine's rivers and streams have been designated for classes of protection equal to or higher than Maine's interpretation of the CWA Interim Goal (i.e., Class C). Reclassification upgrades have been implemented with strong public and legislative support. The state has designated water bodies into higher classes to protect waters currently demonstrating high quality and to retain improvements in lower quality waters that had been restored to higher conditions due to wastewater treatment successes. During the nearly three decades since 1987, the Maine State Legislature has assigned 13,955 river and stream miles to a Class A or Class AA management goal, an increase of 25.5%[34]. Numeric biological criteria and articulation of the gradient of aquatic life management classes facilitated the recognition of both the presence of high quality waters and improvements in condition due to remediation. As shown in Figures 21 and 22, these classification upgrades have mostly been drawn from Class B and Class C waters where biological monitoring data demonstrated either the potential, or the actual achievement of the standards of Class A or Class AA. Without their ALU classification approach, TALUs, and criteria, these gains in condition would likely have gone un-detected and unprotected. Additionally, the state's ecologically descriptive condition classes have enhanced public understanding of existing conditions, problems, and restorable target conditions. They provide an important tool in building public and stakeholder support for the often substantial investment that is required to restore aquatic resources.

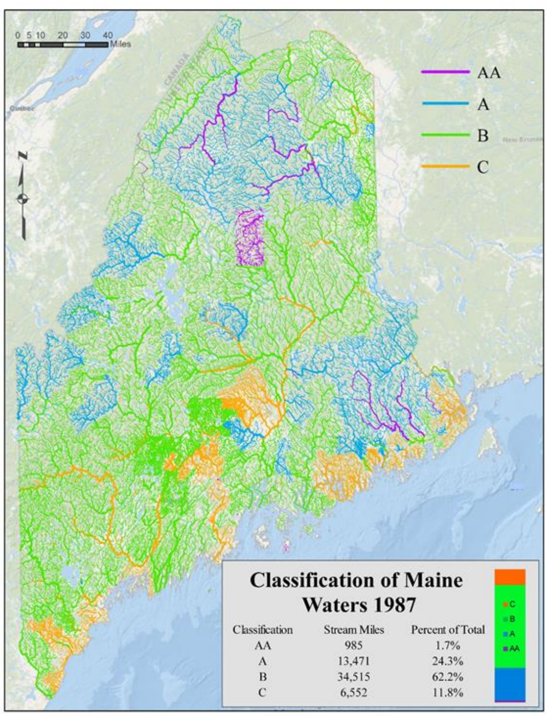


Figure 59. Distribution of Maine water quality classifications in 1987 prior to WQS revisions.

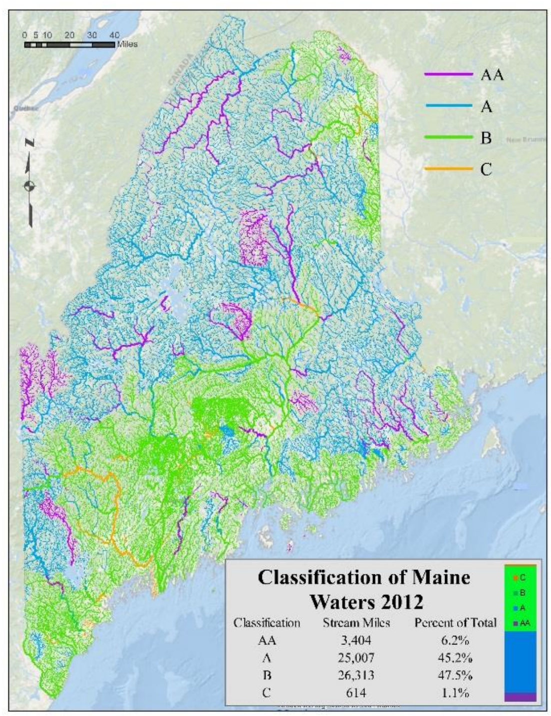


Figure 60. Distribution of Maine water quality classifications in 2012 following 25 years of water quality improvements and classification upgrades.

[34] See State of Maine Water Quality Standards Docket, http://www.maine.gov/dep/water/wqs/docket/index.html (Accessed February 2016) and USEPA, State Tribal and Territorial Standards http://water.epa.gov/scitech/swguidance/standards/wqslibrary/me_index.cfm (Accessed February 2016).

6.5.5 Early Detection and Management of an Emerging Problem

When Maine's Biological Monitoring Program was initiated, a primary concern was management of point source discharges. Implementation of Best Available Technology for point sources eliminated many of these causes of biological impairment with the result that the aquatic life in receiving waters throughout the State rebounded to significantly improved conditions (Davies et al. 1999; Davies et al. In press). More recently, however, biological assessment of smaller streams has revealed impairment caused by changes in physical stream conditions (e.g., increased impervious surfaces in the watershed, hydrologic and stream channel shape alteration). Chemical assessments in these smaller streams have documented increased nutrients and toxic constituent concentrations, salt runoff, increased temperature, and decreased DO.

In 2006, Maine became one of the first states to issue TMDLs based on the percent of a stream watershed covered by impervious surfaces such as roads and parking lots (% IC) (Meidel and MEDEP 2006a, 2006b). Narrative and numeric biological criteria in Maine's WQS were used as the TMDL end point, goal, and ultimate numeric water quality compliance measure for the impaired portions of the streams in order to address non-attainment of ALUs. The restoration pathway described in the TMDL focused on realistic approaches to minimizing the biological, physical, and chemical *effects* of impervious cover, rather than direct elimination of IC. Expanding on the success of the 2006 % IC TMDL, in 2012, MEDEP completed a statewide % IC TMDL for 30 urban impaired streams and 5 associated wetlands (MEDEP 2012). As in 2006, the 2012 TMDL also included aquatic life restoration targets based on the relationship of % IC in the stream watersheds and target improvements in macroinvertebrate community condition.

In 2015, MEDEP conducted a fine-scale geospatial analysis of % IC in watersheds upstream of algal and macroinvertebrate biological assessment sites and determined attainment of TALU for each assemblage at those sites (Danielson et al. In press). Watershed % IC estimates were computed in ArcMap with 1-meter, high-resolution spatial data from 2004 and 2007. Results, shown in Figure 61, revealed that in general, streams become vulnerable to no longer attaining Class AA/A biological criteria when % IC in upstream watersheds is in the range of 1%–3% IC. The risk of not attaining Class B biological criteria increases in the range of 3%–6% IC. Finally, the transition from low risk to high risk of attaining Class C criteria is in the range of 10%–15% IC.

The % IC study revealed that small streams are at risk of impairment at lower levels of watershed % IC than previously recognized. Recognizing the difficulty, expense, and extended lag times associated with urban stream restoration, environmental managers and urban planners in Maine increasingly realize the importance and cost-effectiveness of *preventing* impairment of urban streams. TALU and BCG concepts, along with rigorous biological assessment data, helped MDEP raise awareness about the vulnerability of biological assemblages to urbanization and other human-caused stressors. This information is used in Maine at both the state and local level to inform water quality management decisions and local land use planning and design of development.

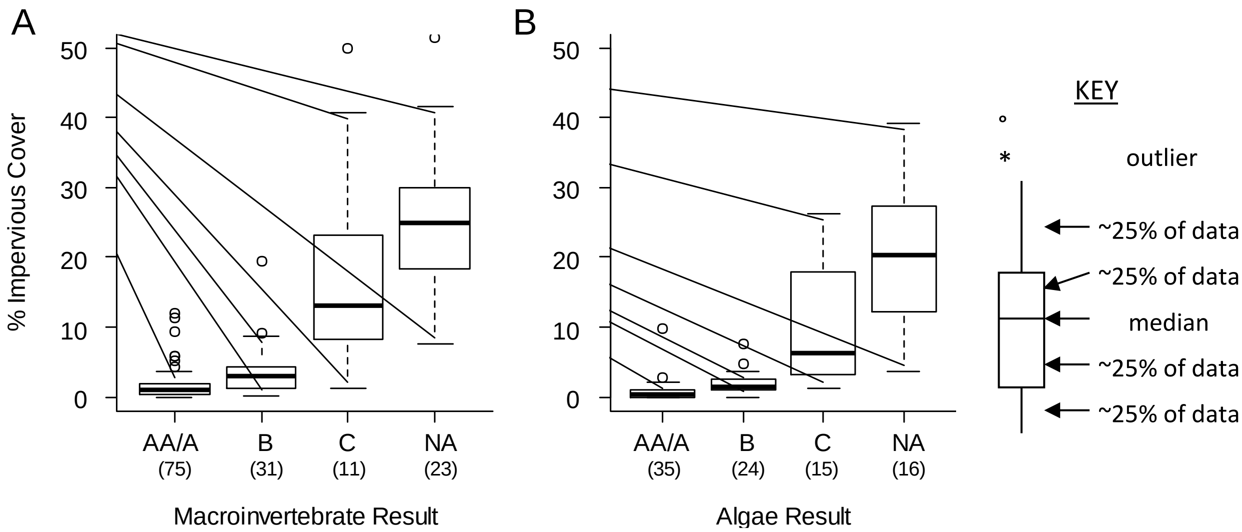
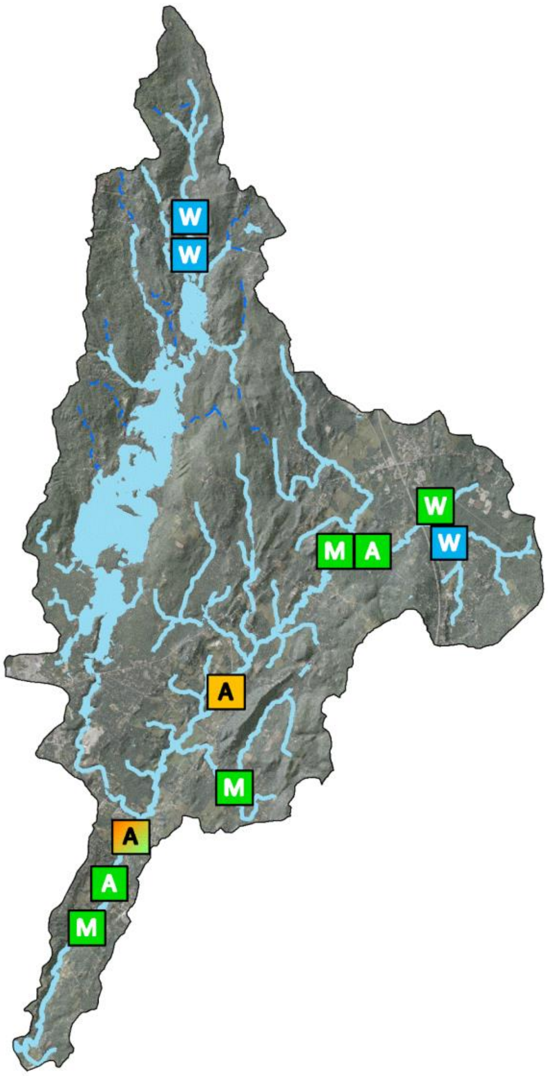


Figure 61. Box-and-whisker plot of % IC of samples grouped by biological assessment results for (A) macroinvertebrates and (B) algae with number of samples in parentheses. The NA group includes samples that do not attain biological criteria for Classes AA/A, B, or C (Source: Danielson et al. In press).

6.5.6 Monitoring and Assessment to Determine Current Condition: Using Biological Condition Gradient Concepts to Integrate Biological Information from Multiple Assemblages and Water Body Types



	WATERBODY	ASSEMBLAGE	CLASS	BCG LEVEL
W	WETLAND	MACROINVERTEBRATE	A	2
W	WETLAND	MACROINVERTEBRATE	B	3
M	STREAM	MACROINVERTEBRATE	B	3
A	STREAM	ALGAE	B	3
A	STREAM	ALGAE	B/C	3/4
A	STREAM	ALGAE	C	4

Figure 62. Pleasant River sites with attained water quality class and BCG level for different assemblages and water body types.

BCG concepts provide Maine with a common assessment framework for comparing biological integrity among different types of water bodies (wetlands, rivers, and streams), regardless of the assemblage assessed or the sampling methods used. This enables MEDEP to evaluate condition and threats to aquatic resources on a watershed basis. The integrated assessment also contributes important information for design of remediation activities, even in the absence of formally promulgated numeric biological criteria. For example, MEDEP evaluated the condition of the Pleasant River watershed using multiple biological assessment models, water quality class attainment, expert judgment, the BCG, and supporting chemical and physical information. Located in southern Maine, the Pleasant River watershed is primarily forested with some agriculture, as well as increasing amounts of urbanization in the downstream portions of the watershed. The Pleasant River has a TALU goal of Class B. MEDEP sampled algae and macroinvertebrates in several locations on the Pleasant River and sampled macroinvertebrates in several headwater wetlands (MEDEP 2006, 2009, 2014; Danielson et al. 2011). Biological assessment showed that the headwater stream and wetland samples attained Class A or B biological criteria using macroinvertebrate data (Figure 62).

However, further downstream, the stream macroinvertebrate samples attained Class B biological criteria, but stream algal samples were mixed, attaining Class B or C. MEDEP used water chemistry data, habitat evaluations, diagnostic algal and macroinvertebrate metrics, expert judgment, and the BCG concept to determine that nutrient pollution was the probable stressor to which the algal community was responding. A watershed survey identified potential sources of nutrients in the lower part of the watershed. The combination of biological assessments for two water body types and taxonomic groups allowed MEDEP to complete a more holistic and meaningful evaluation of the Pleasant River watershed than what could have been accomplished with only one biological assessment method. MEDEP now has a tool to detect early signals of nutrient pollution before the full aquatic community is detrimentally impacted.

Findings from biological assessments of multiple assemblages and water body types have also been used to improve and strengthen Maine's statewide impervious cover TMDL report.[35] For example, in Maine's 2010 Integrated Water Quality Report, Capisic Brook in Portland and Westbrook, Maine was 303(d)-listed for stream benthic macroinvertebrate impairment based on MEDEP's numeric biological criteria rule. Although numeric biological criteria for Maine wetlands had not yet been formally promulgated, Capisic Pond was also listed for wetland macroinvertebrate impairments based on interpretation of quantitative data showing that narrative ALUs were not attained. The state's multivariate biological assessment models for wetland macroinvertebrates and stream algae enabled results to be compared to Maine's TALU classes and macroinvertebrate numeric biological criteria. Stream algal and wetland macroinvertebrate biological assessments helped biologists determine that Capisic Pond and Capisic Brook were not attaining narrative biological criteria, resulting in biological impairment listing for multiple causes.

6.5.7 Using Maine's Tiered Aquatic Life Uses and Biological Assessment Methods to Evaluate Wetland Condition

The MEDEP Biological Monitoring Program assesses the health of inundated emergent and aquatic bed freshwater wetlands. Samples consist of aquatic macroinvertebrates, planktonic and epiphytic algae, and physical and chemical data related to trophic state and habitat condition (MEDEP 2006; MEDEP 2009). Sampling typically occurs in freshwater marshes and fringing wetlands associated with rivers, streams, lakes, and ponds. The biological assessment statistical model for wetlands provides an objective means of assessing condition.

Maine has found that wetland biological assessment provides a complementary approach to assessments of wetland function and value. Under the definitions established by the USEPA *Wetland Core Elements of an Effective State and Tribal Wetlands Program*[36] Maine conducts a "level 3" biological assessment of wetlands. According to EPA, "level 3 or intensive site assessments provide a more thorough and rigorous measure of wetland condition by gathering direct and detailed measurements of biological taxa and/or hydro-geomorphic functions." Maine's wetland macroinvertebrate biological assessment program can detect incremental differences in aquatic resource condition utilizing a locally calibrated statistical model consistent with the BCG concepts (MDEP 2006; MDEP 2009). Additional applications of wetland biological assessments include determining whether wetlands attain designated ALUs, tracking trends over time, and, in conjunction with chemical and physical assessments, diagnosing stressors, and assessing impacts or threats related to land use practices (e.g., point source discharges, toxic contaminants, hydropower, and water withdrawal projects).

In 2013, the MEDEP Biological Monitoring Program evaluated the biological condition of wetland compensatory mitigation projects using wetland biological assessment methods (DiFranco et al. 2013).

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[35] See http://www.maine.gov/dep/water/monitoring/tmdl/tmdl2.html. Accessed February 2016.

[36] See http://water.epa.gov/grants_funding/wetlands/cefintro.cfm. Accessed February 2016.

Mitigating adverse environmental impacts of development is an integral part of Maine's Natural Resources Protection Act,[37] a state law regulating land use activities and administered by MEDEP. The State of Maine or federal agencies administering resource protection regulations might require appropriate and practicable compensatory mitigation as a condition of granting a permit to alter or destroy wetlands. Compensation is defined in the NRPA as "replacement of a lost or degraded wetland function with a function of equal or greater value." If ecologically appropriate compensation is not available or otherwise practicable, a permit applicant may request to pay an *in-lieu* compensation fee to be used for the purpose of restoring, enhancing, creating or preserving other resource functions or values that are environmentally equal or preferable to the functions and values being lost. Upon authorization the In-Lieu Fee is placed in a "Natural Resource Mitigation Fund" administered by The Nature Conservancy's (TNC's) Maine office.

For this study, MEDEP wanted to determine whether compensatory mitigation projects supported aquatic life communities comparable to minimally disturbed reference sites. The MEDEP Biological Monitoring Program evaluated quantitative biological data, biological assessment model results, expert judgment, and the BCG, to compare the biological condition of 9 wetland compensation sites to that of 51 minimally disturbed reference sites. The mitigation sites in the study represented a cross section of available Maine "permittee-responsible" compensation projects that used restoration, creation, enhancement, and preservation techniques, and were completed between 1995 and 2007. The compensation projects varied in age and encompassed a range of freshwater wetland types, including forested, scrub-shrub, emergent, wet meadow, aquatic bed, and open water marsh.

Figure 63 illustrates comparisons of reference and mitigation sites for sensitive versus tolerant taxa metrics using box and whisker plots and quantile (cumulative distribution) plots. In general, mitigation sites had fewer numbers and types of sensitive taxa and a higher proportion of eurytopic taxa (i.e., taxa that are adapted to a wide range of environmental conditions). Table 37 shows estimated BCG condition based on data analysis, expert judgment and the provisional wetland biological assessment model (DiFranco et al. 2013). Results of this study indicated that community structure is significantly different between a set of 51 reference wetlands and nine mitigation wetlands based on taxa tolerance metrics and BCG level. This type of information can improve monitoring and assessment of mitigation sites.

[37] See NRPA, http://www.maine.gov/dep/land/nrpa/index.html (Accessed February 2016), 38 M.R.S.A. § 480 A–BB.

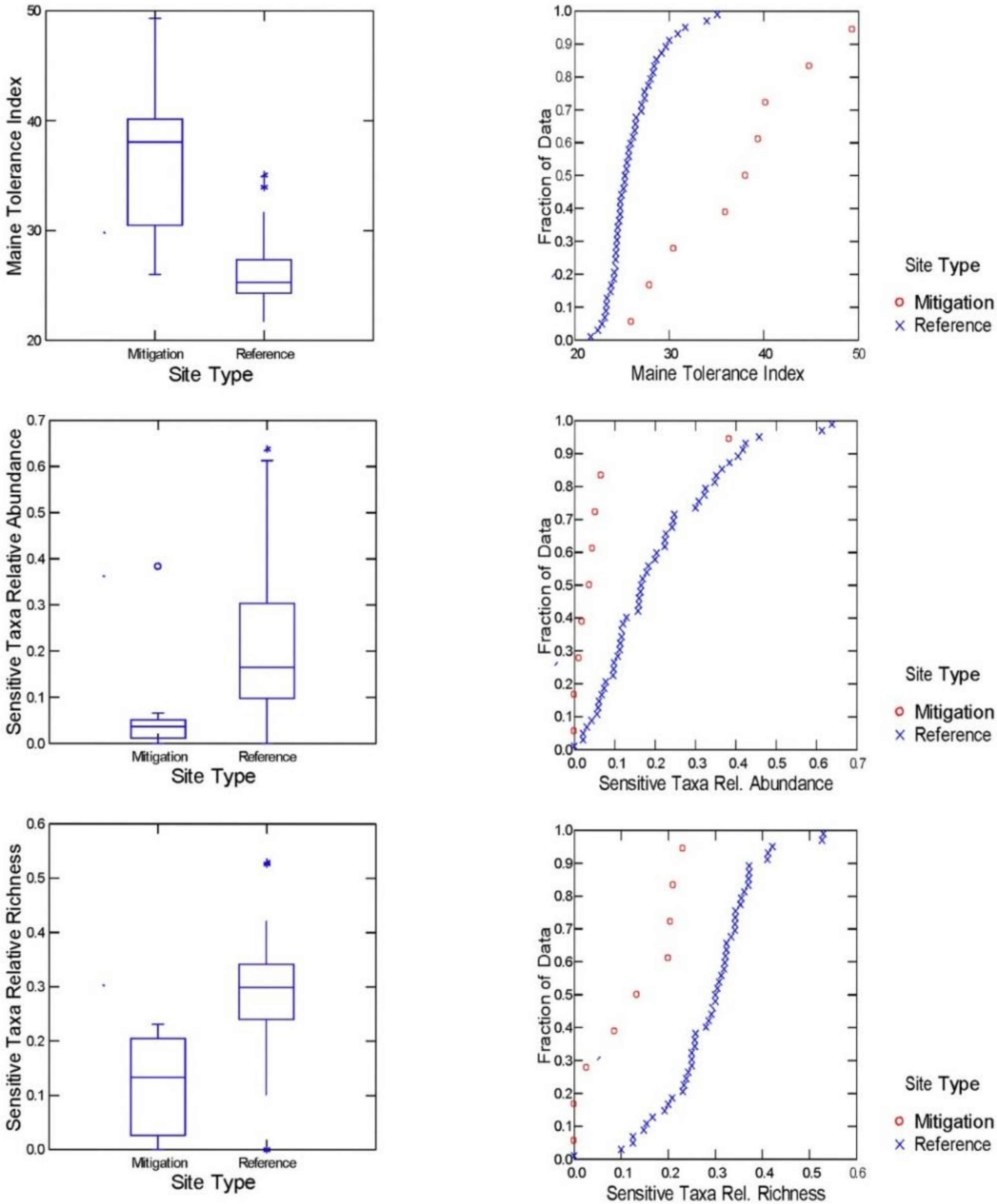


Figure 63. Comparison of reference and mitigation sites for the Maine Tolerance Index and sensitive/tolerant taxa metrics (reference site N=51; mitigation site N=9) (DiFranco et al. 2013).

Table 37. Measured values of chemical and watershed stressors, attained water quality classes, and corresponding BCG levels of reference wetlands and mitigation wetlands (DiFranco et al. 2013)

Mitigation Site Station Number	Specific Conductance µS/cm	Total Phosphorus (mg/L)	MEDEP Human Disturbance Score	% Watershed Alteration	Assigned Legislative Class	BCG Level
Reference site range	**9–95**	**.005–.097**	**1–10**	**0–5.5**		**2.5–4.5**
Reference site mean	**30.6**	**.017**	**5**	**1.9**		**2.8**
W-171	98	0.15	26	24.1	B	5.2[3]
W-173	141	0.22	20	74.7	B	5.5
W-174	57	0.071	10	37.6	C	4.2
W-175	25	0.013	23	16.7	B	4.2
W-179	265	0.051	23	84.0	B	5.5
W-180	76	0.032	22	21.9	B	4.2
W-181	163	0.091	24	39.9	C	4.8
W-182	1120	0.069	40	100	B	4.5
W-184	234	0.027	22	73.3	B	4.5

[1] Reference site classification attainment: Class AA/A or Class B: 78%; Class C: 8%; Non-attainment: 0

[2] Non-attainment of Class C (i.e., lower than the lowest Maine ALU standards)

[3] MEDEP assigns BCG scores utilizing digits to the right of the decimal point to indicate the strength of association, e.g., level 3.2 means "Leans toward level 2"; level 3.5 means "Solid level 3", level 3.8 means "Leans toward level 4".

6.5.8 Conclusion

For Maine, their approach to classifying waters based on current ecological condition provides a direct linkage to CWA biological integrity objectives and ALU goals. This direct linkage facilitates effective communication with stakeholders and water quality management decision makers on current conditions and the likelihood for improvements. As sustained and significant improvements in biological condition were observed based on systematic monitoring of streams, these improvements were documented and class assignments for specific streams were upgraded (e.g., Class C to B; Class B to A as appropriate). As Maine further develops and applies biological assessment tools and data to water bodies other than streams (e.g., wetlands, estuaries, lakes, large rivers), the BCG is included as part of their toolbox.

6.6 Ohio: Use of Biological Gradient to Support Water Quality Management

6.6.1 Key Message

Ohio has used biological assessment information in conjunction with chemical water quality and physical habitat assessments to support water quality management decisions since the late 1970s. While the Ohio ALU classification framework pre-dated the BCG by 25 years, it is based on concepts that are parallel to the BCG, highlighting the relationship between biology, habitat, and the potential for water quality improvements. Ohio's ecological based approach contributed both technical and implementation "lessons learned" to conceptualization of the BCG (Davies and Jackson 2006). The state's biological monitoring and assessment program has provided timely information about the status of individual water bodies and the data to support water quality management program information needs for more than 35 years. This includes when biological conditions improve and when revisions of designated uses are warranted. A systematic process to determine which use(s) is (are) appropriate and attainable for a stream or river has been and remains the key first step in using biological assessment data to support water quality management.

6.6.2 Background

A major aspect of the development of the Ohio biological assessment program and tiered ALU framework is the experience gained through the sustained development of systematic biological assessments beginning in the late 1970s and through the 1980s. This is where the methods, concepts, and theories were tested, applied, and refined, resulting in a tractable system for measuring biological quality at appropriate spatial scales and through time. Qualitative, narrative guidelines were initially used to assess biological status via systematic watershed monitoring and assessment. The data and experiences gained in this early assessment process provided the raw materials for incorporating the concepts of biological integrity that emerged later. Further refinements were also made to the biological assessment tools and the tiered uses including how they are assigned and assessed. Keys to the success of this approach were the initial decisions about indicator assemblages and methods. These have remained stable through time with no major modifications that could have resulted in disconnections within the statewide database that is more than 35 years old.

Ohio EPA formally adopted numeric biological criteria into the Ohio Water Quality Standards (Ohio WQS; Ohio Administrative Code 3745-1) in 1990. The biological criteria have been used to guide and enhance water quality management programs and assess their environmental outcomes. As a result, the state refined definitions of some ALUs, adopted new ones, and added numerical biological criteria to support a tiered approach to water quality management within the Ohio WQS (Table 38). The numeric biological criteria are an outgrowth of an existing framework of TALUs and narrative biological assessment criteria that had been in place since the late 1970s (Table 39 and Table 40). Ohio's approach to biological assessment evolved from an initial reliance on best professional judgment guided by the narrative biological criteria for determining the quality of fish and macroinvertebrate assemblages to a more quantitative and independent approach based on calibrated indices and numeric biological criteria. While the early narrative descriptions of four levels of quality ranging from excellent to poor (Table 39 and Table 40) predated the BCG, the narrative attributes and the rating of multiple levels of condition are consistent with the attributes and scaling of the current BCG. These concepts were retained and further refined with the development of the fish IBI and invertebrate community integrity index (ICI) and the derivation of numeric biological criteria for the current Ohio TALUs (Figure 64) which were initially mapped to the BCG as part of the early BCG development workshops hosted by EPA (Figure 65).

Table 38. Descriptive summary of Ohio's tiered aquatic life use designations

Aquatic Life Use	Key Attributes	Why a Water body Would Be Designated	Practical Impacts (compared to a baseline of WWH)
Warmwater Habitat (WWH)	Balanced assemblages of fish/invertebrates comparable to least impacted *regional* reference condition	Either supports biota consistent with numeric biological criteria for that ecoregion **or** exhibits the habitat potential to support recovery of the aquatic fauna	Baseline regulatory requirements consistent with the CWA "fishable" and "protection & propagation" goals; criteria consistent with EPA guidance with state/regional modifications as appropriate
Exceptional Warmwater Habitat (EWH)	Unique and/or diverse assemblages; comparable to upper quartile of *statewide* reference condition	Attainment of the EWH biological criteria demonstrated by both organism groups	More stringent criteria for DO, temperature, ammonia, and nutrient targets; more stringent restrictions on dissolved metals translators; restrictions on nationwide dredge & fill permits; may result in more stringent wastewater treatment requirements
Coldwater Habitat (CWH)	Sustained presence of Salmonid or non-salmonid coldwater aquatic organisms; bonafide trout fishery	Biological assessment reveals coldwater species as defined by Ohio EPA (2014); put-and-take trout fishery managed by Ohio Department of Natural Resources	Same as above except that common metals criteria are more stringent; may result in more stringent wastewater treatment requirements
Modified Warmwater Habitat (MWH)	Warmwater assemblage dominated by species tolerant of low DO, excessive nutrients, siltation, and/or habitat modifications	Impairment of the WWH biological criteria; existence and/or maintenance of hydrological modifications that cannot or will not be reversed or abated In the foreseeable future so that WWH biological criteria can be attained; a UAA is required	Less stringent criteria for DO, ammonia, and nutrient targets; less restrictive applications of dissolved metals translators; Nationwide permits apply without restrictions or exception; may result in less restrictive wastewater treatment requirements
Limited Resource Waters (LRW)	Highly degraded assemblages dominated exclusively by tolerant species; *should not* reflect acutely toxic conditions	Extensive physical and hydrological modifications that cannot be reversed, are essentially irretrievable and which preclude attainment of higher uses; a UAA is required	Chemical criteria are based on the prevention of acutely lethal conditions; may result in less restrictive wastewater treatment requirements

Table 39. Narrative biological criteria (fish) for determining ALU designations and attainment of CWA goals (November, 1980; after Ohio EPA 1981)

Evaluation Class Category	"Exceptional" Class I (EWH)	"Good" Class II (WWH)	"Fair" Class III	"Poor" Class IV
1.	Exceptional or unusual assemblage of species	Usual association of expected species	Some expected species absent, or in very low abundance	Most expected species absent
2.	Sensitive species abundant	Sensitive species present	Sensitive species absent, or in very low abundance	Sensitive species absent
3.	Exceptionally high diversity	High diversity	Declining diversity	Low diversity
4.	Composite index > 9.0–9.5	Composite index > 7.0–7.5; < 9.0–9.5	Composite index > 4.5–5.0; < 7.0–7.5	Composite index < 4.0–4.5
5.	Outstanding recreational Fishery		Tolerant species increasing, beginning to dominate	Tolerant species dominate
6.	Rare, endangered, or threatened species present			

Conditions: Categories 1, 2, 3, and 4 (if data are available) must be met and 5 or 6 must also be met in order to be

Table 40. Narrative biological criteria (macroinvertebrates) for determining ALU designations and attainment of CWA goals (November 1980; after Ohio EPA 1981)

Evaluation Class Category	"Exceptional" Class I (EWH)	"Good" Class II (WWH)	"Fair" Class III	"Poor" Class IV designated in a particular class.
1.	Pollution sensitive species abundant	Pollution sensitive species present in moderate numbers	Pollution sensitive species present in low numbers	Pollution sensitive species absent
2.	Intermediate species present in low numbers	Intermediate species present in moderate numbers	Intermediate species abundant	Intermediate species present in low numbers or absent
3.	Tolerant species present in low numbers	Tolerant species present in low numbers	Tolerant species present in moderate numbers	Tolerant species abundant (all types may be absent if extreme toxic conditions exist)
4.	Number of taxa > 30[1]	Number of taxa 25–30	Number of taxa 20–25	Number of taxa < 20
5.	Exceptional diversity Shannon index < 3.5	High diversity Shannon index 2.9–3.5	Moderate diversity Shannon index 2.3–2.9	Low diversity Shannon index < 2.3

[1]Number of quantitative taxa from artificial substrates.

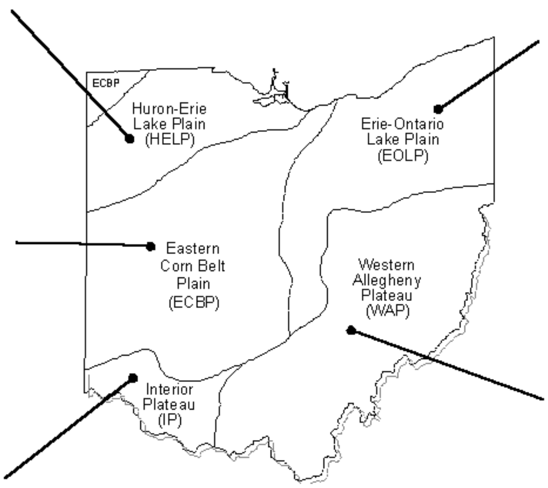
Ohio Biological Criteria: Adopted May 1990 (OAC 3745-1-07; Table 7-14)

Huron Erie Lake Plain (HELP)

Use	Size	IBI	MIwb	ICI
WWH	H	28	NA	34
	W	32	7.3	34
	B	34	8.6	34
MWH-C	H	20	NA	22
	W	22	5.6	22
	B	20	5.7	22
MWH-I	B	30	5.7	NA

Erie Ontario Lake Plain (EOLP)

Use	Size	IBI	MIwb	ICI
WWH	H	40	NA	34
	W	38	7.9	34
	B	40	8.7	34
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-I	B	30	6.6	NA



Eastern Corn Belt Plains (ECBP)

Use	Size	IBI	MIwb	ICI
WWH	H	40	NA	36
	W	40	8.3	36
	B	42	8.5	36
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-I	B	30	6.6	NA

Western Allegheny Plateau (WAP)

Use	Size	IBI	MIwb	ICI
WWH	H	44	NA	34
	W	44	8.4	34
	B	40	8.6	34
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-A	H	24	NA	30
	W	24	5.5	30
	B	24	5.5	30
MWH-I	B	30	6.6	NA

Interior Plateau (IP)

Use	Size	IBI	MIwb	ICI
WWH	H	40	NA	30
	W	40	8.1	30
	B	38	8.7	30
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-I	B	30	6.6	NA

Statewide Exceptional Criteria

Use	.Size	IBI	MIwb	ICI
EWH	H	50	NA	46
	W	50	9.4	46
	B	48	9.6	46

Figure 64. Numeric biological criteria adopted by Ohio EPA in 1990, showing stratification of biological criteria by biological assemblage, index, site type, ecoregion for warmwater and modified warmwater habitat (WWH and MWH, respectively), and statewide for the exceptional warmwater habitat (EWH) use designations.

Developed and adopted by Ohio EPA in 1978, the original tiered aquatic life use narratives represented a major revision to a general use framework that was adopted in 1974. Ohio's tiered uses recognized the different types of warmwater aquatic assemblages that corresponded to the mosaic of natural features of the landscape and nearly two centuries of human-induced changes. The eventual development of more refined tiered uses and numeric biological criteria that are in place today was the result of sustained state support to develop a biological monitoring and assessment program with technical capability to discriminate incremental changes in biological condition with increasing stress. The empirical evidence used to develop the initial concepts for tiered uses can be found in comprehensive works on the natural history and zoogeography of the Midwest such as *Fishes of Ohio* (Trautman 1957, 1981). This and other natural history texts documented the natural and human-caused variations in the distribution, composition, and abundance of biological assemblages over space and through time including before and after European settlement. Trautman (1957) not only provides a detailed narrative of Ohio's natural history, but describes the biological evidence that was used to formulate the initial concepts about biological integrity that emerged in the late 1970s and early 1980s and which were later incorporated in the BCG. Such works also described the key features of the landscape that influence and determine the potential aquatic fauna of water bodies and were the forerunners of the regionalization frameworks that appeared soon after. As an alternative to a "one-size-fits-all" approach, these provided an important foundation for the development of Ohio's tiered uses. The articulation of a practical definition of biological integrity by Karr and Dudley (1981) provided a theoretical framework for the development of Ohio's numeric biological criteria (Figure 65). Key components of this framework are: (1) using biological assemblages as a direct measure of ALU attainment status (Herricks and Schaeffer 1985; Karr et al. 1986), (2) the development and use of IBIs as assessment tools (Karr 1981; Karr et al. 1986), (3) derivation of regional reference condition to determine appropriate and attainable ALU goals and assessment endpoints (Hughes et al. 1986), and (4) systematic monitoring and assessment of the state's rivers and streams using a pollution survey design. These represented a major advancement over previous attempts (Ballantine and Guarria 1975) to define and develop a workable framework to address the concept of biological integrity. Embedded in this framework is the recognition that water quality management must be approached from an ecological perspective that is grounded in sound ecological theory *and* which is validated by empirical observation. This means developing monitoring and assessment and WQS to encompass the five factors that determine the integrity of a water resource Figure 22; Karr et al. 1986).

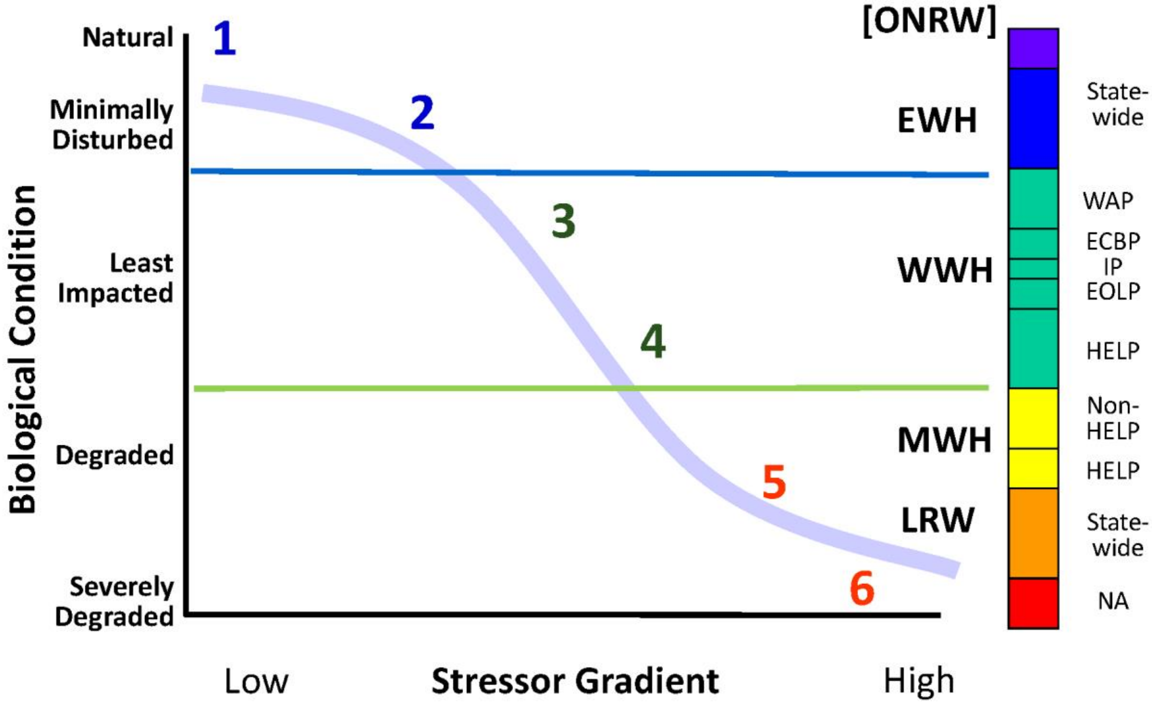
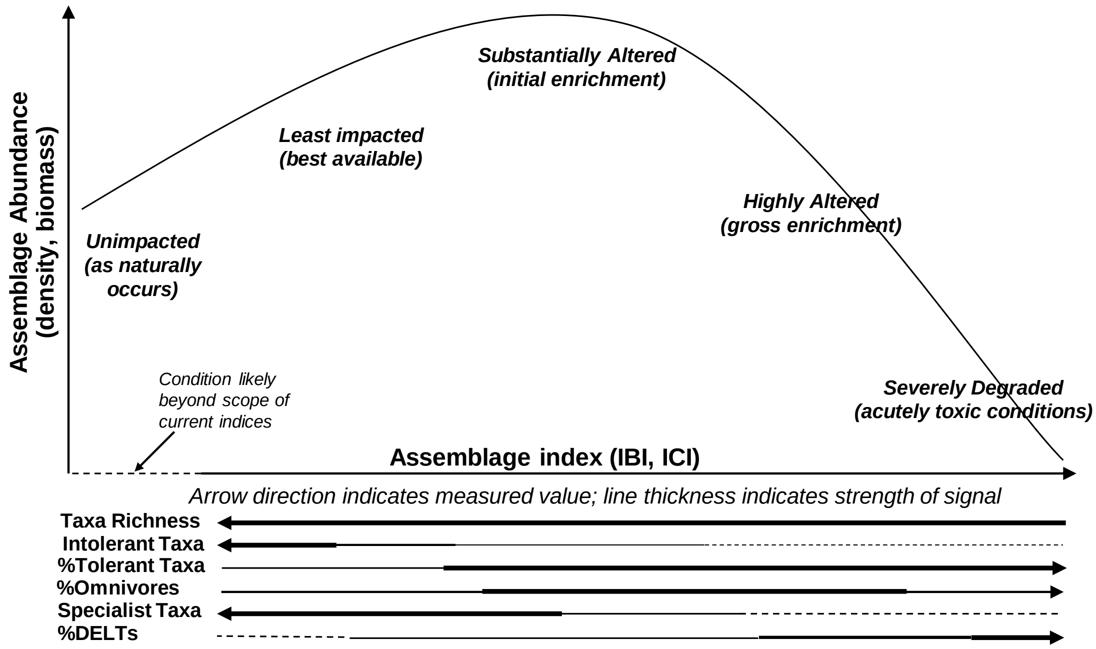


Figure 65. An initial mapping of the Ohio TALUs to the BCG relating descriptions of condition along the y1-axis and ranges of condition encompassed by the numerical biological criteria for each of four tiered use subcategories and the highest antidegradation tier (ONRW) along the y2-axis. ONRW – Outstanding National Resource Waters; EWH – Exceptional Warmwater Habitat; WWH – Warmwater Habitat; MWH – Modified Warmwater Habitat; LRW – Limited Resource Waters.

The understanding of fish and macroinvertebrate assemblage responses to stressor gradients ranging from minimally disturbed to severely altered conditions was affirmed by repeated empirical observations of assemblage responses which are depicted in Figure 66. This graphic represents measured assemblage abundance (y-axis) against assemblage indices (fish IBI, macroinvertebrate ICI; x-axis) with the response of selected metrics and other assemblage attributes at increments along what



Biological and Stressor Gradient Descriptors

"As Naturally Occurs" *(Pristine)*	"Least Impacted" *(Exceptional)*	"Initial Enrichment" *(Good)*	"Moderate Enrichment" *(Fair)*	"Gross Enrichment" *(Poor)*	"Severely Degraded" *(Very Poor)*
Assemblage Characteristics					
Native assemblages; no symptoms of stress	"Best of what's left" assemblages; high richness; intolerants, specialists predominate	"Typical" assemblages; good richness; emerging symptoms of stress in selected metrics	"Impaired" assemblages; tolerants & generalists predominate numbers/bio-mass; loss of intolerants	"Degraded"; highly tolerant taxa pre-dominate; reduced abundance; anomalies increasing	"Severely degraded"; very low numbers; few taxa; very high % anomalies; toxic signatures
Chemical Water Quality Conditions					
As natural; no human-made compounds present	"Best reference" quality; toxics < detection; high D.O., low nutrients	"Background reference" quality; toxics < chronic; adequate D.O., nutrients = reference	"Enriched" quality; toxics < chronic; marginal D.O. regime, nutrients > reference	"Degraded" quality; toxics > chronic; low D.O., nutrients **>>** reference	"Extremely poor" quality; toxics ≥ acute; very low D.O., nutrients **>>** reference; contaminated sediments
Physical Habitat & Flow Regime					
Natural habitat and flow regime; no human-made modifications	Excellent quality habitat & flow regime; recovered from human-made modifications	Good quality habitat & flow regime; *de minimis* human modifications	Fair quality habitat & flow regime; active human modifi-cations; incomplete recovery	Poor quality habitat & flow regime; active human modifi-cations; no recovery	Severe modifi-cations; ephemeral flows; active human modifications; no recovery potential
Examples of Sources and Activities					
No effects of human activity are evident	Point sources present, do not dominate flows; NPS impacts buffered by extensive riparian system	Point sources may dominate flows; NPS impacts buffered by good riparian zones	PS/NPS enrich-ment impacts; NPS unbuffered; channel modifi-cations; im-poundments	Gross PS/NPS enrichment impacts inc. CSOs; NPS unbufferd; chan-nel modifications; urbanization	Severe PS/NPS toxic impacts; extreme channel modifications; urbanization; acid mine drainage, severe thermal

Figure 66. Descriptive model of the response of fish and macroinvertebrate assemblage metrics and characteristics to a quality gradient and different levels of impact from stressors in Midwestern U.S. warmwater rivers and streams (modified from Ohio EPA 1987 and Yoder and Rankin 1995b).

is in reality a continuum. Biological descriptions correspond to the six levels of the then emerging BCG model and include descriptions of key assemblage characteristics, chemical water quality conditions, physical habitat and flow regime, and sources of stress that are typically associated with each. This was modified from the original conceptual model of Ohio EPA (1987a) and Yoder and Rankin (1995b), and it includes the probable upper limits of Ohio's fish and macroinvertebrate indices. It demonstrates that understanding the relationship between assemblage responses and stressors is a fundamental aspect of using biological assessments to support condition assessments *and* water quality management programs. It also demonstrates the pre-BCG concepts that eventually merged in the formal development and description of the current BCG.

6.6.3 Determining Appropriate Levels of Protection

By merging the ALU framework with systematic monitoring and assessment, Ohio has been able to determine attainable levels of condition for streams and rivers and also to set protection levels for high quality waters. This framework is consistently applied within a rotating basin sequence of "biological surveys" that address the following questions:

1) Is the current designated ALU appropriate and attainable and if not, what is the appropriate use for a water body?
2) Are the biological criteria for the most appropriate and attainable use tier attained?
3) Have there been any changes through time and what do they portend for water quality management?

The scale of monitoring and assessment is sufficiently detailed so that designations of individual water bodies or segments of a water body can be made based on scientific information and data. Getting this task done correctly affects everything that follows including assessments of condition and which WQS will guide water quality management actions such as permitting and TMDLs. The data gathered by a biological survey is processed, evaluated, and synthesized in a biological and water quality report. The report serves as the rationale for justifying recommended changes to a currently assigned ALU. The report also identifies sources of pollutants and/or pollution contributing to impairment(s) of the recommended designated uses. The recommendations for use designation revisions are a direct output of the biological and water quality assessment. Recommended revisions to the WQS are based on a UAA framework that emphasizes the demonstrated *potential* to attain a particular use tier based on the following information (and in order of importance):

1) Attainment of the numeric biological criteria for WWH[38] or EWH results in designation of that use; or,
2) If the WWH biological criteria are not attained, the habitat determined by the Qualitative Habitat Evaluation Index (QHEI; Rankin 1995) based on an assessment of habitat attributes is used to determine the *potential* to attain WWH.

[38] WWH – Warmwater Habitat is the minimum condition that meets the 101[a][2] goal of the Clean Water Act under the Ohio WQS. A UAA is required to designate a river or stream to a lower use (e.g., MWH or LRW).

For uses below WWH (i.e., MWH or LRW), a UAA is performed and includes consideration of the restorability of the water body and of the factors that may preclude WWH attainment. This process requires the following information:

1) The current attainment status of the water body based on a biological assessment performed in accordance with the requirements of the biological criteria, the Ohio WQS, and the Five-Year Monitoring Strategy;
2) A habitat assessment to evaluate the potential to attain WWH; and,
3) A reasonable relationship between the impaired status and the precluding human-caused activities based on an assessment of multiple indicators used in their most appropriate indicator roles and a demonstration consistent with 40 CFR Part 131.10[g].

Since 1978 Ohio EPA has used a consistent process to validate and, if necessary, revise uses in the Ohio WQS. The codified uses for approximately 2,000 streams and rivers have been revised using this process (Figure 67) and information from a biological and water quality assessment. This became a routine practice once the assessment criteria and decision making process for UAAs were established in the mid-1980s. It required the parallel development of reliable tools, particularly for determining status, assessing habitat, and determining causal associations, all of which is part of the developmental process described in several documents and publications (Ohio EPA 1987; 2006; Rankin 1989; 1995; Yoder 1995). The terms "upgrade" and "downgrade" are used only as descriptions of the direction of change from the current codified use to that derived from systematic monitoring and assessment. The vast majority of these changes are from the baseline of original designations that were made in 1978 without the benefit of systematic monitoring and assessment data, numerical biological criteria, and refinements in the process that occurred in the mid-1980s. Hence, these original designations are merely being replaced by the most appropriate use designation based on consistently applied criteria and assessments. Undesignated streams are almost always smaller watersheds of < 5–10 mi^2 drainage area that were missed by the default stream naming format that was employed when stream and river specific designations were originally adopted in 1985. Prior to that time, smaller tributaries were "automatically" assigned the use tier of the parent mainstem river or stream, a practice that resulted in numerous erroneous use designations. The more frequent monitoring of these smaller streams and watersheds in the 1990s and 2000s was partially the result of a shift in emphasis to watershed based TMDLs which resulted in numerous undesignated streams being monitored and hence designated for the first time. A detailed fact sheet is prepared for each use designation rulemaking to communicate the types of proposed changes to the WQS, the rationale for the changes, and which rivers and streams are affected by the proposed changes. When use designation rulemakings are underway, fact sheets specific to affected river basins can be found on Ohio EPA's website.[39]

[39] See http://epa.ohio.gov/dsw/dswrules.aspx#120473212-early-stakeholder-outreach. Accessed February 2016.

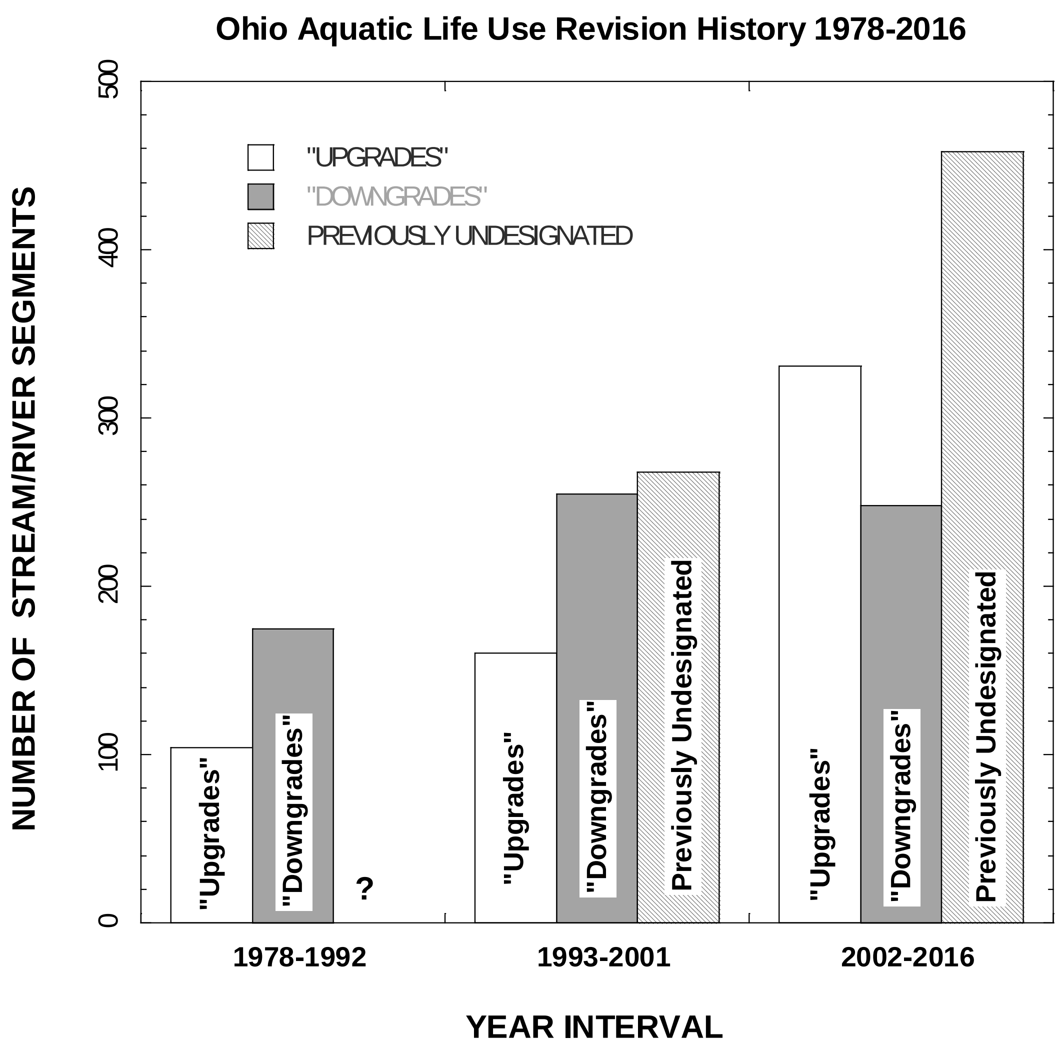


Figure 67. The number of individual stream and river segments in which ALU designations were revised during 1978–1992, 1993–2001, and 2002–2016. Cases where the use was revised to a higher use are termed "upgrades" and cases where a lower use was assigned are termed "downgrades." Previously undesignated refers to streams that were not listed in the 1985 WQS, but which were added as each was designated as a result of systematic monitoring and assessment. The number of waters previously undesignated in the first interval is unknown.

The Ohio tiered use and biological criteria framework and their application to Ohio rivers and streams were first tested in the Ohio court system in 1989 and were validated by a lower court and upheld in appeals up to, and including, the Ohio Supreme Court (NEORSD vs. Shank No. 89-1554, Supreme Court of Ohio, Feb. 27, 1991). The application of the biological criteria to justify additional pollution controls in response to a biological impairment was likewise validated by a lower court and upheld in subsequent appeals (City of Salem vs. Korleski No. 09AP-620, Tenth District Court of Appeals, March 23, 2010; Ohio Supreme Court 2010-0818; appeal not accepted, August 25, 2010).

6.6.4 Setting Attainable Goals for Improvements

Ecologically-based tiered uses, a systematic approach to monitoring and assessment, and a tractable UAA process can provide substantial benefits for water quality management programs related to guiding efforts to improve conditions and assessing the effectiveness of those efforts in protecting and restoring an ALU. The identification of the recovery potential for aquatic life in a water body using a systematic approach can help set attainable goals for improvements and support evaluation of environmental risks. The Ohio case example illustrates the role of tiered ALUs using a BCG approach for interpretation of conditions, systematic monitoring and assessment, and a consistent process for conducting UAAs in support of TMDLs. The UAA process is routinely applied as a result of the systematic monitoring and assessment of Ohio rivers and streams (Figure 68). The data are used to support recommendations for revisions to the Ohio WQS on an annual basis.

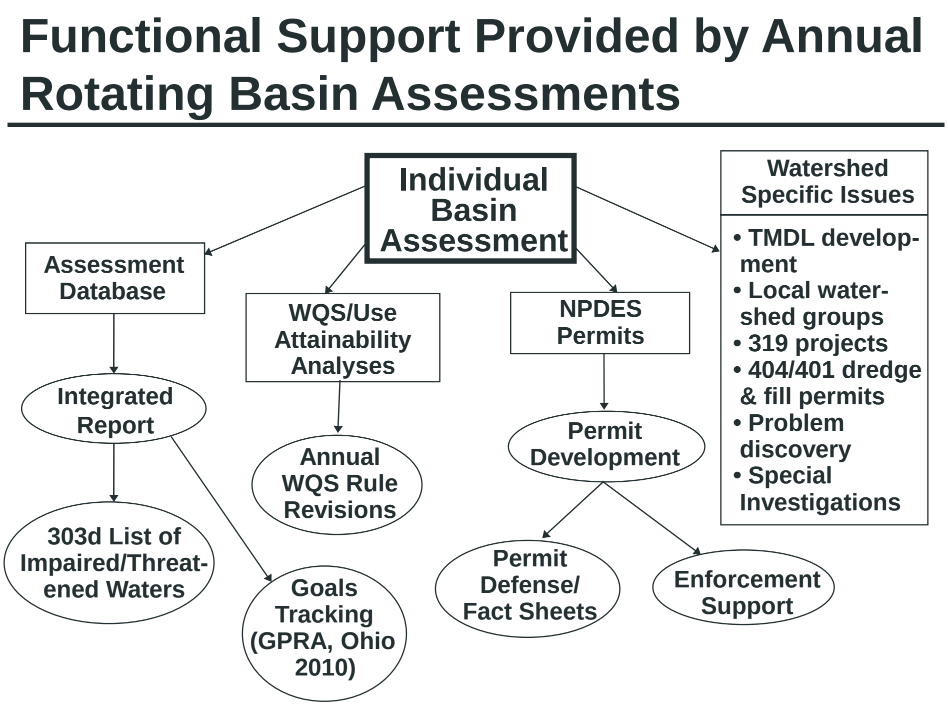


Figure 68. The flow of information from biological and water quality assessments to support for major water quality management programs in Ohio.

Ohio's tiered ALU designation procedures were incorporated into the TMDL process beginning in 1999 (Figure 69; Ohio EPA 1999). Figure 69 illustrates the steps for validating the most appropriate tiered ALU and then basing a TMDL on the criteria embodied by that use tier and the attendant assessment of the receiving streams and rivers. It also illustrates the delineation of the severity and extent of impairments, the most probable causes of the impairments, and follow-up assessments to validate TMDL effectiveness. Because the Ohio EPA monitoring and assessment strategy includes chemical, physical, and biological indicators which are used in their most appropriate roles as indicators of stress, exposure, and response (Yoder and Rankin 1998), support for the development of TMDLs can go beyond addressing singular pollutants to addressing the combination of pollution and pollutants that impair an ALU.

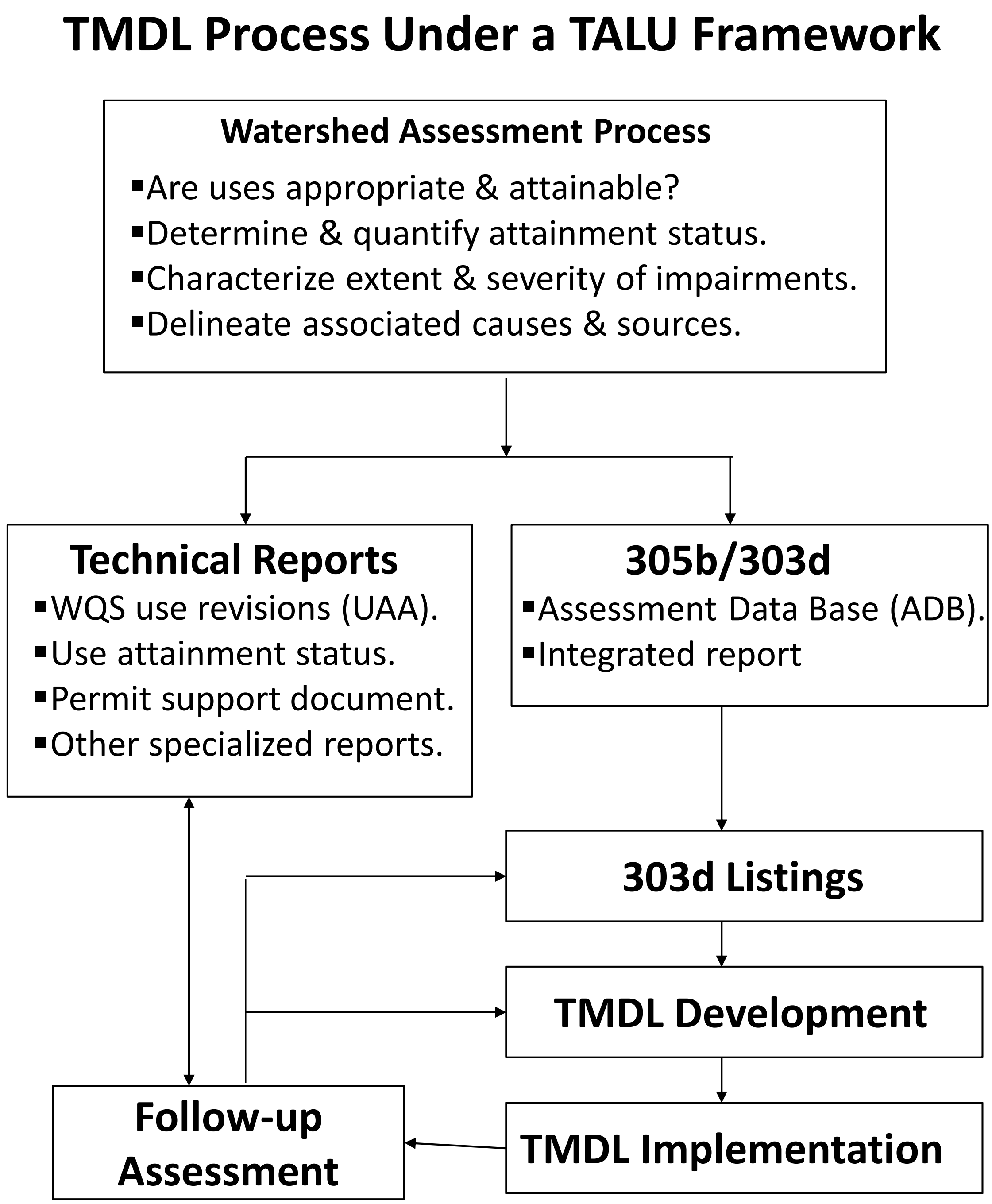


Figure 69. Key steps showing how a TALU based framework can be used to organize and guide a TMDL development and implementation process.

6.6.5 Protecting High Quality Water Bodies

Ohio's antidegradation rule (Ohio Administrative Code 3745-1-05) incorporates levels of protection between the minimum required under the CWA and the maximum protection afforded by federal regulations. The most stringent application of antidegradation is to disallow any lowering of water quality in waters listed as ONRWs. The minimum requirement allows for a lowering of water quality to the minimum WQS applicable to the water body if a determination is made that lowering water quality is necessary to accommodate important social and economic development. However, lowering of water quality below that which is necessary to protect an existing use is prohibited. Ohio has two intermediate levels of protection for certain ecologically important water bodies that permanently reserve a portion of the unused pollutant assimilative capacity, thereby assuring maintenance of a water quality that is better than that prescribed by the prevailing designated use tier. The two intermediate levels are: (1) Outstanding State Water (OSW; Figure 70), and (2) Superior High Quality Water (SHQW) which fall in between ONRW and General High Quality Waters (GHQWs; Figure 71). High quality water bodies are valued public resources because of their ecological and human benefits. Their biological components act as an early warning system that can indicate potential threats to human health, degradation of aesthetic values, reductions in the quality and quantity of recreational opportunities, and other ecosystem

Figure 70. The Mohican River in northeastern Ohio—a candidate for OSW classification because of its high quality ecological and recreational attributes.

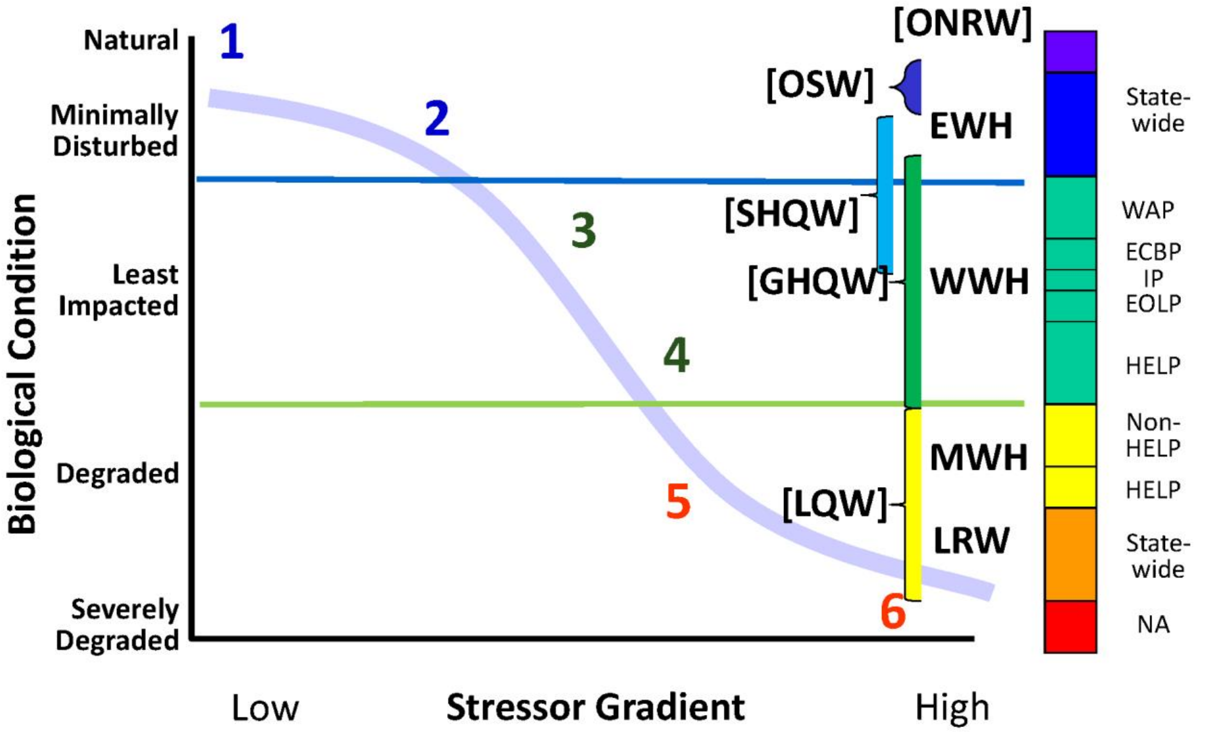


Figure 71. Mapping the Ohio antidegradation tiers to the BCG relating descriptions of condition along the y1-axis and ranges of condition encompassed by the numerical biological criteria for each of four tiered use subcategories and the four antidegradation tiers along the y2-axis. ONRW – Outstanding National Resource Waters; OSW – Outstanding State Waters; SHQW – Superior High Quality Waters; GHQW – Generally High Quality Waters; LQW – Low Quality Waters; EWH – Exceptional Warmwater Habitat; WWH – Warmwater Habitat; MWH – Modified Warmwater Habitat; LRW – Limited Resource Waters.

benefits, or services. The ability of streams and rivers to provide these beneficial services and to act as environmental sentinels is reduced whenever their integrity is degraded. Under the Ohio antidegradation rule, a portion of the remaining assimilative capacity is reserved for water bodies classified as OSW or SHQW in order to preserve an already existing high quality.

Ohio uses a number of biological and physical attributes to place river and stream segments into the OSW, SHQW, and GHQW antidegradation tiers (Table 41). Included are the presence of state or federally listed endangered and threatened species, declining fish species (as defined in the antidegradation rules), the fish and macroinvertebrate assemblage indices (IBI and ICI), the QHEI, the vulnerability of the river or stream to increased stressors, the relative abundance of fish species sensitive to pollution and habitat destruction, and the accumulation of multiple attributes. Adjustments are also made for the Lake Erie drainage to account for the fewer endemic fish and mussel species. Additional considerations include other designations, such as state and national scenic river status, outstanding biodiversity among all aquatic assemblages, exceptionally high quality habitat, and the presence of unique landforms along geological and geomorphological boundaries.

Table 41. General guidelines for nominating OSW, SHQW, and GHQW categories in Ohio. Attributes are considered both singly and in the aggregate

Attribute	OSW	SHQW	GHQW

Attribute	OSW	SHQW	GHQW
Endangered & Threatened Species	Multiple species; large populations; include the most vulnerable species	Present, smaller populations; may include less vulnerable species	Absent or, if present, small populations or of low vulnerability
Declining Fish Species	> 4 declining fish species/segment; large populations	2–4 declining fish species/segment; moderate populations	< 2 declining fish species/segment; typically small populations
IBI and ICI	High mean scores; very high max scores (> 56)	Lower mean scores; fewer high max scores or, if more high scores, few other attributes	Lower mean scores; few or no very high max scores
QHEI	High percentage of QHEI scores ≥ 80	Fewer QHEI scores ≥ 80, many above 70	Few or no QHEI scores ≥ 80, fewer above 70
Vulnerability	Little wastewater effluent; high vulnerability	May be more wastewater effluent; moderate vulnerability	Lower vulnerability; for vulnerable components, antidegradation application may still be denied
Relative Abundance of Fish Species Sensitive to Pollution and Habitat Destruction	Relative abundance is ≥ 3 standard deviations compared to statewide collections of similar sized streams	Relative abundance is ≥ 2 standard deviations compared to statewide collections of similar sized streams	Relative abundance is < 2 standard deviations compared to statewide collections of similar sized streams
Multiple Attributes	High co-occurrence of above attributes	Lower co-occurrence of above attributes or individual attributes more marginal	Little co-occurrence of above attributes, individual attributes often marginal if present

6.6.6 Conclusion

The Ohio approach to classifying waters based on current ecological condition and potential for improvement provides a direct linkage to the CWA biological Integrity objective and ALU goals. This direct linkage enables more effective communication with stakeholders and water quality management decision makers on current conditions and likelihood for improvements. The BCG-like approach enables Ohio EPA to account for biological expectations relative to ecoregion and drainage area and provides a numeric value that synthesizes everything that is being experienced by the biota that can be tracked, monitored, and compared over time to determine if conditions are improving, stabilizing, or deteriorating. As chemical, physical, and biological monitoring has been coordinated and the database expanded, critical information for investigating cause and source of biological impairments has been built and has enabled water quality managers to target sources of stressors and their mechanism of action on the aquatic ecosystem. Because of this database, the state has been able to develop water quality goals for some parameters less well-suited to the classic dose-response relationship for DO and many toxicants. Ohio's ecologically-based approach to classifying waters combined with a robust monitoring program has provided a scientifically defensible method to categorize waters into designated uses and antidegradation tiers. The process has generated UAAs and justification documents as an accepted and routine rulemaking process, primarily resulting in incremental upgrades as controls and BMPs were implemented and improvements observed.

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Glossary

aquatic assemblage	An association of interacting populations of organisms in a given water body; for example, fish assemblage or a benthic macroinvertebrate assemblage.
aquatic community	An association of interacting assemblages in a water body, the biotic component of an ecosystem.
aquatic life use	A beneficial use designation in which the water body provides, for example, suitable habitat for survival and reproduction of desirable fish, shellfish, and other aquatic organisms.
attribute	The measurable part or process of a biological system.
benthic macroinvertebrates or benthos	Animals without backbones, living in or on the sediments, of a size large enough to be seen by the unaided eye and which can be retained by a U.S. Standard no. 30 sieve (28 meshes per inch, 0.595-mm openings); also referred to as benthos, infauna, or macrobenthos.
best management practice	An engineered structure or management activity, or combination of those, that eliminates or reduces an adverse environmental effect of a pollutant.
biological assessment or bioassessment	An evaluation of the biological condition of a water body using surveys of the structure and function of a community of resident biota.
biological criteria or biocriteria	Narrative expressions or numeric values of the biological characteristics of aquatic communities based on appropriate reference conditions; as such, biological criteria serve as an index of aquatic community health.
biological indicator or bioindicator	An organism, species, assemblage, or community characteristic of a particular habitat, or indicative of a particular set of environmental conditions.
biological integrity	The ability of an aquatic ecosystem to support and maintain a balanced, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats in a region.
biological monitoring or biomonitoring	Use of a biological entity as a detector and its response as a measure to determine environmental conditions; ambient biological surveys and toxicity tests are common biological monitoring methods.
biological survey or biosurvey	Collecting, processing, and analyzing a representative portion of the resident aquatic community to determine its structural and/or functional characteristics.

biotope	An area that is relatively uniform in physical structure and that is identified by a dominant biota.
catchment	An incremental watershed that drains directly into a stream reach and excludes upstream areas.
Clean Water Act	The act passed by the U.S. Congress to control water pollution (formally referred to as the Federal Water Pollution Control Act of 1972). Public Law 92-500, as amended. 33 U.S.C. 1251 *et seq*.
Clean Water Act section 303(d)	This section of the act requires states, territories, and authorized tribes to develop lists of impaired waters for which applicable WQS are not being met, even after point sources of pollution have installed the minimum required levels of pollution control technology. The law requires that the jurisdictions establish priority rankings for waters on the lists and develop TMDLs for the waters. States, territories, and authorized tribes are to submit their lists of waters on April 1 in every even-numbered year.
Clean Water Act section 305(b)	Biennial reporting requires description of the quality of the nation's surface waters, evaluation of progress made in maintaining and restoring water quality, and description of the extent of remaining problems.
Clean Water Act section 304(a) criteria	EPA-published, recommended water quality criteria that consist of scientific information regarding concentrations of specific chemicals or levels of parameters in water that protect aquatic life and human health. The States may use these contents as the basis for developing enforceable water quality standards.
criteria	Elements of state water quality standards, expressed as constituent concentrations, levels, or narrative statements, representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use.
designated uses	Those uses specified in WQS for each water body or segment whether or not they are being attained.
disturbance	Human activity that alters the natural state and can occur at or across many spatial and temporal scales.
ecological integrity	The condition of an unimpaired ecosystem as measured by combined chemical, physical (including physical habitat), and biological attributes. Ecosystems have integrity when they have their native components (plants, animals and other organisms) and processes (such as growth and reproduction) intact.

ecoregion	A relatively homogeneous ecological area defined by similarity of climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variables.
function	Processes required for normal performance of a biological system (may be applied to any level of biological organization).
guild	A group of organisms that exhibit similar habitat requirements and that respond in a similar way to changes in their environment.
historical data	Data sets from previous studies, which can range from handwritten field notes to published journal articles.
index of biological/biotic integrity	An integrative expression of site condition across multiple metrics; an IBI is often composed of at least seven metrics.
invasive species	A species whose presence in the environment causes economic or environmental harm or harm to human health. Native species or nonnative species can show invasive traits, although that is rare for native species and relatively common for nonnative species. (Note that this term is not included in the biological condition gradient [BCG].)
least disturbed condition	The best available existing conditions with regard to physical, chemical, and biological characteristics or attributes of a water body within a class or region. Such waters have the least amount of human disturbance in comparison to others in the water body class, region, or basin. Least disturbed conditions can be readily found but can depart significantly from natural, undisturbed conditions or minimally disturbed conditions. Least disturbed condition can change significantly over time as human disturbances change.
maintenance of populations	Sustained population persistence; associated with locally successful reproduction and growth.
metric	A calculated term or enumeration that represents some aspect of biological assemblage, function, or other measurable aspect and is a characteristic of the biota that changes in some predictable way with increased human influence.
minimally disturbed condition	The physical, chemical, and biological conditions of a water body with very limited, or minimal, human disturbance.
multimetric index	An index that combines indicators, or metrics, into a single index value. Each metric is tested and calibrated to a scale and transformed into a unitless score before being aggregated into a multimetric index. Both the index and metrics are useful in assessing and diagnosing ecological condition. See **index of biological/biotic integrity (IBI).**

narrative biological criteria	Written statements describing the structure and function of aquatic communities in a water body that support a designated aquatic life use.
native	An original or indigenous inhabitant of a region; naturally present.
nonnative or intentionally introduced species	With respect to an ecosystem, any species that is not found in that ecosystem; species introduced or spread from one region of the United States to another outside their normal range are nonnative or non-indigenous, as are species introduced from other continents.
numeric biological criteria	Specific quantitative measures of the structure and function of aquatic communities in a water body necessary to protect a designated aquatic life use.
periphyton	A broad organismal assemblage composed of attached algae, bacteria, their secretions, associated detritus, and various species of microinvertebrates.
rapid bioassessment protocols	Cost-effective techniques used to survey and evaluate the aquatic community to detect aquatic life impairments and their relative severity.
rebuttable presumption	In the context of water quality standards, the concept that the CWA 101(a)(2) uses are attainable and therefore must be assigned to a water body, unless a State or Tribe affirmatively demonstrates, with appropriate documentation, that such uses are not attainable.
recovery potential	In the context of water quality management, the likelihood that an impaired water body can be restored so that it ultimately meets water quality standards. Consideration of ecological, stressor, and social factors are involved in the consideration of recovery potential.

reference condition (biological integrity)	The condition that approximates natural, unaffected conditions (biological, chemical, physical, and such) for a water body. Reference condition (biological integrity) is best determined by collecting measurements at a number of sites in a similar water body class or region undisturbed by human activity, if they exist. Because undisturbed conditions can be difficult or impossible to find, minimally or least disturbed conditions, combined with historical information, models, or other methods can be used to approximate reference condition as long as the departure from natural or ideal is understood. Reference condition is used as a benchmark to determine how much other water bodies depart from this condition because of human disturbance. See definitions for minimally and least disturbed condition
reference site	A site selected for comparison with sites being assessed. The type of site selected and the types of comparative measures used will vary with the purpose of the comparisons. For the purposes of assessing the ecological condition of sites, a reference site is a specific locality on a water body that is undisturbed or minimally disturbed and is representative of the expected ecological integrity of other localities on the same water body or nearby water bodies.
refugia	Accessible microhabitats or regions in a stream reach or watershed where adequate conditions for organism survival are maintained during circumstances that threaten survival; for example, drought, flood, temperature extremes, increased chemical stressors, habitat disturbance.
sensitive taxa	Taxa intolerant to a given anthropogenic stress; first species affected by the specific stressor to which they are *sensitive* and the last to recover following restoration.
sensitive or regionally endemic taxa	Taxa with restricted, geographically isolated distribution patterns (occurring only in a locale as opposed to a region), often because of unique life history requirements. Can be long-lived, late-maturing, low-fecundity, limited-mobility, or require mutualist relation with other species. Can be among listed endangered/threatened or special concern species. Predictability of occurrence often low; therefore, requires documented observation. Recorded occurrence can be highly dependent on sample methods, site selection, and level of effort.

sensitive-rare taxa	Taxa that naturally occur in low numbers relative to total population density but can make up large relative proportion of richness. Can be ubiquitous in occurrence or can be restricted to certain micro-habitats, but because of low density, recorded occurrence is dependent on sample effort. Often stenothermic (having a narrow range of thermal tolerance) or coldwater obligates; commonly K-strategists (populations maintained at a fairly constant level; slower development; longer life span). Can have specialized food resource needs or feeding strategies. Generally intolerant to significant alteration of the physical or chemical environment; are often the first taxa observed to be lost from a community.
sensitive-ubiquitous taxa	Taxa ordinarily common and abundant in natural communities when conventional sample methods are used. Often having a broader range of thermal tolerance than sensitive or rare taxa. These are taxa that constitute a substantial portion of natural communities and that often exhibit negative response (loss of population, richness) at mild pollution loads or habitat alteration.
stressors	Physical, chemical, and biological factors that adversely affect aquatic organisms.
structure	Taxonomic and quantitative attributes of an assemblage or community, including species richness and relative abundance structurally and functionally redundant attributes of the system and characteristics, qualities, or processes that are represented or performed by more than one entity in a biological system.
taxa	A grouping of organisms given a formal taxonomic name such as species, genus, family, and the like.
taxa of intermediate tolerance	Taxa that compose a substantial portion of natural communities; can be r-strategists (early colonizers with rapid turnover times; boom/bust population characteristics). Can be eurythermal (having a broad thermal tolerance range). Can have generalist or facultative feeding strategies enabling utilization of relatively more diversified food types. Readily collected with conventional sample methods. Can increase in number in waters with moderately increased organic resources and reduced competition but are intolerant of excessive pollution loads or habitat alteration.

tolerant taxa	Taxa that compose a small proportion of natural communities. They are often tolerant of a broader range of environmental conditions and are thus resistant to a variety of pollution- or habitat-induced stresses. They can increase in number (sometimes greatly) in the absence of competition. Commonly r-strategists (early colonizers with rapid turnover times; boom/bust population characteristics), able to capitalize when stress conditions occur; last survivors.
total maximum daily load	The sum of the allowable loads of a single pollutant from all contributing point and nonpoint sources; the calculated maximum amount of a pollutant a water body can receive and still meet WQS and an allocation of that amount to the pollutant's source.
water quality management (nonregulatory)	Decisions on management activities relevant to a water resource, such as problem identification, need for and placement of best management practices, pollution abatement actions, and effectiveness of program activity.
water quality standard	A law or regulation that consists of the designated use or uses of a water body, the narrative or numerical water quality criteria (including biological criteria) that are necessary to protect the use or uses of that water body, and antidegradation requirements.
whole effluent toxicity	The aggregate toxic effect of an aqueous sample (e.g., whole effluent wastewater discharge) as measured by an organism's response after exposure to the sample (e.g., lethality, impaired growth or reproduction); WET tests replicate the total effect and actual environmental exposure of aquatic life to toxic pollutants in an effluent without requiring the identification of the specific pollutants.

Abbreviations and Acronyms

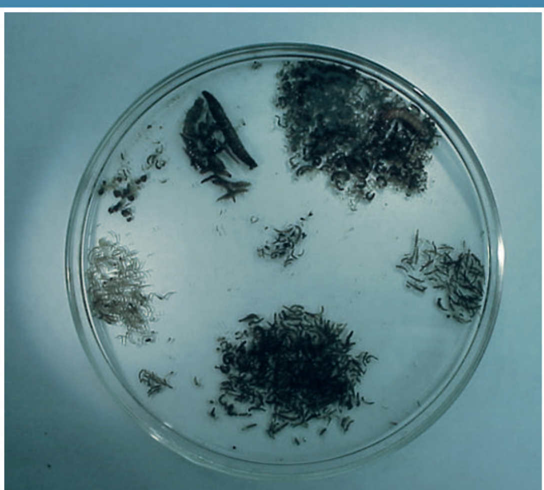
ADEM	Alabama Department of Environmental Management
AIS	aquatic invasive species
ALAWADR	Alabama Water-Quality Assessment and Monitoring Data Repository
ALU	aquatic life use
ANOVA	univariate analysis of variance
aRPD	apparent redox potential discontinuity
ATtiLA	Analytical Tools Interface for Landscape Assessments
AUSRIVAS	AUStralian RIVer Assessment System
BCG	biological condition gradient
BEAST	BEnthic Assessment of SedimenT
BMP	best management practice
BT	brook trout
CADDIS	Causal Analysis/Diagnosis Decision Information System
CART	classification and regression tree (statistical analysis)
CBP	Chesapeake Bay Program
CCA	Canonical Correspondence Analysis
CFR	*Code of Federal Regulations*
CIBI	Continuous Index of Biological Integrity
CNMI	Commonwealth of the Northern Mariana Islands
CRW	Coral Reef Watch, NOAA
CT DEEP	Connecticut Department of Energy and Environmental Protection
Cu	copper
CWA	Clean Water Act
CWH	coldwater habitat
DELT	deformities, erosion, lesions, and tumors
D-IBI	diadromous index of biotic integrity
DO	dissolved oxygen
EDAS	Environmental Data Acquisition System
EMAP	Environmental Monitoring and Assessment Program
EPA	U.S. Environmental Protection Agency
EPT	ephemeroptera, plecoptera, trichoptera taxa
ESD	environmental site design
E/T	endangered/threatened
EV	exceptional value
FACI	Fish Assessment Community Index
F-IBI	fish index of biological/biotic integrity
GAM	general additive model
GHQW	General High Quality Water

GLEI	Great Lakes Environmental Indicators
GRE	Great Rivers Evaluation
GRFIn	Great River Fish Index
GSA	generalized stress axis
HDG	human disturbance gradient
HDS	human disturbance score
HQ	high-quality
HUC	hydrologic unit code
HWI	Healthy Watershed Index
IBI	index of biological/biotic integrity
IC	impervious cover
ICI	invertebrate community integrity index
LDI	landscape development intensity index
LDM	linear discriminant model
LRBOI	Little River Band of Ottawa Indians
LRW	limited resource water
LWD	large, woody debris
MANOVA	multivariate analysis of variance
MCDEP	Montgomery County Department of Environmental Protection
MEDEP	Maine Department of Environmental Protection
M-IBI	macroinvertebrate index of biological/biotic integrity
mIBI	modified index of biological integrity
MIwb	modified index of well-being
MMI	multimetric index
M-NCPPC	Maryland-National Capital Park and Planning Commission
MPCA	Minnesota Pollution Control Agency
MWH	modified warmwater habitat
NA	non-attainment
NBEP	Narragansett Bay Estuary Program
NCRMP	National Coral Reef Monitoring Program
NELP	New England large rivers
NHD	National Hydrography Dataset
NIH	National Institutes of Health
NJ DEP	New Jersey Department of Environmental Protection
NLCD	National Land Cover Database
NMDS	non-metric multidimensional scaling
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NRC	National Research Council
O/E	observed over expected
OM	organic matter

ONRW	Outstanding National Resource Water
OSI	Organism-Sediment Index
OSW	Outstanding State Water
PA DEP	Pennsylvania Department of Environmental Protection
PAR	photosynthetic active radiation
PCA	Principal Component Analysis
QHEI	qualitative habitat evaluation index
POM	particulate organic matter
REMAP	Regional Environmental Monitoring and Assessment Program
RIDEM	Rhode Island Department of Environmental Management
RIVPACS	River Invertebrate Prediction and Classification System
RM	river mile
RPS	Recovery Potential Screening
SHQW	Superior High Quality Water
SPI	sediment profile imagery
SST	sea surface temperature
STORET	STOrage and RETrieval
TALU	tiered aquatic life use
TBEP	Tampa Bay Estuary Program
TITAN	Threshold Indicator Taxa ANalysis
TIV	Tolerance Indicator Value
TMC	Ten Mile Creek
TMDL	Total Maximum Daily Load
TNC	The Nature Conservancy
UAA	use attainability analysis
UMRBA	Upper Mississippi River Basin Association
UMR	Upper Mississippi River
USVI	U.S. Virgin Islands
WDG	watershed disturbance gradient
WSIO	Watershed Index Online
WQS	water quality standards
WQTF	Water Quality Task Force
WQV	Weighted Stressor Value
WWH	warmwater habitat
WWTF	wastewater treatment facility

EPA 842-R-16-001

photos courtesy of Maine Department of Environmental Protection


among State agencies and with local and Federal agencies.

(v) Procedures for program management and administration including provision of program financing, training and technical assistance, public participation, and emergency management.

(d) *Indian Tribes.* An Indian Tribe is eligible for the purposes of this rule and the Clean Water Act assistance programs under 40 CFR part 35, subparts A and H if:

(1) The Indian Tribe has a governing body carrying out substantial governmental duties and powers;

(2) The functions to be exercised by the Indian Tribe pertain to the management and protection of water resources which are held by an Indian Tribe, held by the United States in trust for Indians, held by a member of an Indian Tribe if such property interest is subject to a trust restriction on alienation, or otherwise within the borders of an Indian reservation; and

(3) The Indian Tribe is reasonably expected to be capable, in the Regional Administrator's judgment, of carrying out the functions to be exercised in a manner consistent with the terms and purposes of the Clean Water Act and applicable regulations.

(e) *Update and certification.* State and/or areawide agency WQM plans shall be updated as needed to reflect changing water quality conditions, results of implementation actions, new requirements or to remove conditions in prior conditional or partial plan approvals. Regional Administrators may require that State WQM plans be updated as needed. State Continuing Planning Processes (CPPs) shall specify the process and schedule used to revise WQM plans. The State shall ensure that State and areawide WQM plans together include all necessary plan elements and that such plans are consistent with one another. The Governor or the Governor's designee shall certify by letter to the Regional Administrator for EPA approval that WQM plan updates are consistent with all other parts of the plan. The certification may be contained in the annual State work program.

(f) *Consistency.* Construction grant and permit decisions must be made in accordance with certified and approved WQM plans as described in §§ 130.12(a) and 130.12(b).

[50 FR 1779, Jan. 11, 1985, as amended at 54 FR 14360, Apr. 11, 1989; 59 FR 13818, Mar. 23, 1994]

§ 130.7 Total maximum daily loads (TMDL) and individual water quality-based effluent limitations.

(a) *General.* The process for identifying water quality limited segments still requiring wasteload allocations, load allocations and total maximum daily loads (WLAs/LAs and TMDLs), setting priorities for developing these loads; establishing these loads for segments identified, including water quality monitoring, modeling, data analysis, calculation methods, and list of pollutants to be regulated; submitting the State's list of segments identified, priority ranking, and loads established (WLAs/LAs/TMDLs) to EPA for approval; incorporating the approved loads into the State's WQM plans and NPDES permits; and involving the public, affected dischargers, designated areawide agencies, and local governments in this process shall be clearly described in the State Continuing Planning Process (CPP).

(b) Identification and priority setting for water quality-limited segments still requiring TMDLs.

(1) Each State shall identify those water quality-limited segments still requiring TMDLs within its boundaries for which:

(i) Technology-based effluent limitations required by sections 301(b), 306, 307, or other sections of the Act;

(ii) More stringent effluent limitations (including prohibitions) required by either State or local authority preserved by section 510 of the Act, or Federal authority (law, regulation, or treaty); and

(iii) Other pollution control requirements (e.g., best management practices) required by local, State, or Federal authority are not stringent enough to implement any water quality standards (WQS) applicable to such waters.

(2) Each State shall also identify on the same list developed under paragraph (b)(1) of this section those water

quality-limited segments still requiring TMDLs or parts thereof within its boundaries for which controls on thermal discharges under section 301 or State or local requirements are not stringent enough to assure protection and propagation of a balanced indigenous population of shellfish, fish and wildlife.

(3) For the purposes of listing waters under §130.7(b), the term ''water quality standard applicable to such waters'' and ''applicable water quality standards'' refer to those water quality standards established under section 303 of the Act, including numeric criteria, narrative criteria, waterbody uses, and antidegradation requirements.

(4) The list required under §§130.7(b)(1) and 130.7(b)(2) of this section shall include a priority ranking for all listed water quality-limited segments still requiring TMDLs, taking into account the severity of the pollution and the uses to be made of such waters and shall identify the pollutants causing or expected to cause violations of the applicable water quality standards. The priority ranking shall specifically include the identification of waters targeted for TMDL development in the next two years.

(5) Each State shall assemble and evaluate all existing and readily available water quality-related data and information to develop the list required by §§130.7(b)(1) and 130.7(b)(2). At a minimum ''all existing and readily available water quality-related data and information'' includes but is not limited to all of the existing and readily available data and information about the following categories of waters:

(i) Waters identified by the State in its most recent section 305(b) report as ''partially meeting'' or ''not meeting'' designated uses or as ''threatened'';

(ii) Waters for which dilution calculations or predictive models indicate nonattainment of applicable water quality standards;

(iii) Waters for which water quality problems have been reported by local, state, or federal agencies; members of the public; or academic institutions. These organizations and groups should be actively solicited for research they may be conducting or reporting. For example, university researchers, the United States Department of Agriculture, the National Oceanic and Atmospheric Administration, the United States Geological Survey, and the United States Fish and Wildlife Service are good sources of field data; and

(iv) Waters identified by the State as impaired or threatened in a nonpoint assessment submitted to EPA under section 319 of the CWA or in any updates of the assessment.

(6) Each State shall provide documentation to the Regional Administrator to support the State's determination to list or not to list its waters as required by §§130.7(b)(1) and 130.7(b)(2). This documentation shall be submitted to the Regional Administrator together with the list required by §§130.7(b)(1) and 130.7(b)(2) and shall include at a minimum:

(i) A description of the methodology used to develop the list; and

(ii) A description of the data and information used to identify waters, including a description of the data and information used by the State as required by §130.7(b)(5); and

(iii) A rationale for any decision to not use any existing and readily available data and information for any one of the categories of waters as described in §130.7(b)(5); and

(iv) Any other reasonable information requested by the Regional Administrator. Upon request by the Regional Administrator, each State must demonstrate good cause for not including a water or waters on the list. Good cause includes, but is not limited to, more recent or accurate data; more sophisticated water quality modeling; flaws in the original analysis that led to the water being listed in the categories in §130.7(b)(5); or changes in conditions, e.g., new control equipment, or elimination of discharges.

(c) Development of TMDLs and individual water quality based effluent limitations.

(1) Each State shall establish TMDLs for the water quality limited segments identified in paragraph (b)(1) of this section, and in accordance with the priority ranking. For pollutants other than heat, TMDLs shall be established

at levels necessary to attain and maintain the applicable narrative and numerical WQS with seasonal variations and a margin of safety which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality. Determinations of TMDLs shall take into account critical conditions for stream flow, loading, and water quality parameters.

(i) TMDLs may be established using a pollutant-by-pollutant or biomonitoring approach. In many cases both techniques may be needed. Site-specific information should be used wherever possible.

(ii) TMDLs shall be established for all pollutants preventing or expected to prevent attainment of water quality standards as identified pursuant to paragraph (b)(1) of this section. Calculations to establish TMDLs shall be subject to public review as defined in the State CPP.

(2) Each State shall estimate for the water quality limited segments still requiring TMDLs identified in paragraph (b)(2) of this section, the total maximum daily thermal load which cannot be exceeded in order to assure protection and propagation of a balanced, indigenous population of shellfish, fish and wildlife. Such estimates shall take into account the normal water temperatures, flow rates, seasonal variations, existing sources of heat input, and the dissipative capacity of the identified waters or parts thereof. Such estimates shall include a calculation of the maximum heat input that can be made into each such part and shall include a margin of safety which takes into account any lack of knowledge concerning the development of thermal water quality criteria for protection and propagation of a balanced, indigenous population of shellfish, fish and wildlife in the identified waters or parts thereof.

(d) *Submission and EPA approval.* (1) Each State shall submit biennially to the Regional Administrator beginning in 1992 the list of waters, pollutants causing impairment, and the priority ranking including waters targeted for TMDL development within the next two years as required under paragraph (b) of this section. For the 1992 biennial submission, these lists are due no later than October 22, 1992. Thereafter, each State shall submit to EPA lists required under paragraph (b) of this section on April 1 of every even-numbered year. For the year 2000 submission, a State must submit a list required under paragraph (b) of this section only if a court order or consent decree, or commitment in a settlement agreement dated prior to January 1, 2000, expressly requires EPA to take action related to that State's year 2000 list. For the year 2002 submission, a State must submit a list required under paragraph (b) of this section by October 1, 2002, unless a court order, consent decree or commitment in a settlement agreement expressly requires EPA to take an action related to that State's 2002 list prior to October 1, 2002, in which case, the State must submit a list by April 1, 2002. The list of waters may be submitted as part of the State's biennial water quality report required by §130.8 of this part and section 305(b) of the CWA or submitted under separate cover. All WLAs/LAs and TMDLs established under paragraph (c) for water quality limited segments shall continue to be submitted to EPA for review and approval. Schedules for submission of TMDLs shall be determined by the Regional Administrator and the State.

(2) The Regional Administrator shall either approve or disapprove such listing and loadings not later than 30 days after the date of submission. The Regional Administrator shall approve a list developed under §130.7(b) that is submitted after the effective date of this rule only if it meets the requirements of §130.7(b). If the Regional Administrator approves such listing and loadings, the State shall incorporate them into its current WQM plan. If the Regional Administrator disapproves such listing and loadings, he shall, not later than 30 days after the date of such disapproval, identify such waters in such State and establish such loads for such waters as determined necessary to implement applicable WQS. The Regional Administrator shall promptly issue a public notice seeking comment on such listing and loadings. After considering public comment and

making any revisions he deems appropriate, the Regional Administrator shall transmit the listing and loads to the State, which shall incorporate them into its current WQM plan.

(e) For the specific purpose of developing information and as resources allow, each State shall identify all segments within its boundaries which it has not identified under paragraph (b) of this section and estimate for such waters the TMDLs with seasonal variations and margins of safety, for those pollutants which the Regional Administrator identifies under section 304(a)(2) as suitable for such calculation and for thermal discharges, at a level that would assure protection and propagation of a balanced indigenous population of fish, shellfish and wildlife. However, there is no requirement for such loads to be submitted to EPA for approval, and establishing TMDLs for those waters identified in paragraph (b) of this section shall be given higher priority.

[50 FR 1779, Jan. 11, 1985, as amended at 57 FR 33049, July 24, 1992; 65 FR 17170, Mar. 31, 2000; 66 FR 53048, Oct. 18, 2001]

§ 130.8 Water quality report.

(a) Each State shall prepare and submit biennially to the Regional Administrator a water quality report in accordance with section 305(b) of the Act. The water quality report serves as the primary assessment of State water quality. Based upon the water quality data and problems identified in the 305(b) report, States develop water quality management (WQM) plan elements to help direct all subsequent control activities. Water quality problems identified in the 305(b) report should be analyzed through water quality management planning leading to the development of alternative controls and procedures for problems identified in the latest 305(b) report. States may also use the 305(b) report to describe ground-water quality and to guide development of ground-water plans and programs. Water quality problems identified in the 305(b) report should be emphasized and reflected in the State's WQM plan and annual work program under sections 106 and 205(j) of the Clean Water Act.

(b) Each such report shall include but is not limited to the following:

(1) A description of the water quality of all waters of the United States and the extent to which the quality of waters provides for the protection and propagation of a balanced population of shellfish, fish, and wildlife and allows recreational activities in and on the water.

(2) An estimate of the extent to which CWA control programs have improved water quality or will improve water quality for the purposes of paragraph (b)(1) of this section, and recommendations for future actions necessary and identifications of waters needing action.

(3) An estimate of the environmental, economic and social costs and benefits needed to achieve the objectives of the CWA and an estimate of the date of such achievement.

(4) A description of the nature and extent of nonpoint source pollution and recommendations of programs needed to control each category of nonpoint sources, including an estimate of implementation costs.

(5) An assessment of the water quality of all publicly owned lakes, including the status and trends of such water quality as specified in section 314(a)(1) of the Clean Water Act.

(c) States may include a description of the nature and extent of ground-water pollution and recommendations of State plans or programs needed to maintain or improve ground-water quality.

(d) In the years in which it is prepared the biennial section 305(b) report satisfies the requirement for the annual water quality report under section 205(j). In years when the 305(b) report is not required, the State may satisfy the annual section 205(j) report requirement by certifying that the most recently submitted section 305(b) report is current or by supplying an update of the sections of the most recently submitted section 305(b) report which require updating.

[50 FR 1779, Jan.11, 1985, as amended at 57 FR 33050, July 24, 1992]

(iii) An identification of the surface waters for which the Tribe proposes to establish water quality standards.

(4) A narrative statement describing the capability of the Indian Tribe to administer an effective water quality standards program. The narrative statement should include:

(i) A description of the Indian Tribe's previous management experience which may include the administration of programs and services authorized by the Indian Self-Determination and Education Assistance Act (25 U.S.C. 450 *et seq.*), the Indian Mineral Development Act (25 U.S.C. 2101 *et seq.*), or the Indian Sanitation Facility Construction Activity Act (42 U.S.C. 2004a);

(ii) A list of existing environmental or public health programs administered by the Tribal governing body and copies of related Tribal laws, policies, and regulations;

(iii) A description of the entity (or entities) which exercise the executive, legislative, and judicial functions of the Tribal government;

(iv) A description of the existing, or proposed, agency of the Indian Tribe which will assume primary responsibility for establishing, reviewing, implementing and revising water quality standards;

(v) A description of the technical and administrative capabilities of the staff to administer and manage an effective water quality standards program or a plan which proposes how the Tribe will acquire additional administrative and technical expertise. The plan must address how the Tribe will obtain the funds to acquire the administrative and technical expertise.

(5) Additional documentation required by the Regional Administrator which, in the judgment of the Regional Administrator, is necessary to support a Tribal application.

(6) Where the Tribe has previously qualified for eligibility or ''treatment as a state'' under a Clean Water Act or Safe Drinking Water Act program, the Tribe need only provide the required information which has not been submitted in a previous application.

(c) Procedure for processing an Indian Tribe's application.

(1) The Regional Administrator shall process an application of an Indian Tribe submitted pursuant to § 131.8(b) in a timely manner. He shall promptly notify the Indian Tribe of receipt of the application.

(2) Within 30 days after receipt of the Indian Tribe's application the Regional Administrator shall provide appropriate notice. Notice shall:

(i) Include information on the substance and basis of the Tribe's assertion of authority to regulate the quality of reservation waters; and

(ii) Be provided to all appropriate governmental entities.

(3) The Regional Administrator shall provide 30 days for comments to be submitted on the Tribal application. Comments shall be limited to the Tribe's assertion of authority.

(4) If a Tribe's asserted authority is subject to a competing or conflicting claim, the Regional Administrator, after due consideration, and in consideration of other comments received, shall determine whether the Tribe has adequately demonstrated that it meets the requirements of § 131.8(a)(3).

(5) Where the Regional Administrator determines that a Tribe meets the requirements of this section, he shall promptly provide written notification to the Indian Tribe that the Tribe is authorized to administer the Water Quality Standards program.

[56 FR 64895, Dec. 12, 1991, as amended at 59 FR 64344, Dec. 14, 1994]

Subpart B—Establishment of Water Quality Standards

§ 131.10 Designation of uses.

(a) Each State must specify appropriate water uses to be achieved and protected. The classification of the waters of the State must take into consideration the use and value of water for public water supplies, protection and propagation of fish, shellfish and wildlife, recreation in and on the water, agricultural, industrial, and other purposes including navigation. In no case shall a State adopt waste transport or waste assimilation as a designated use for any waters of the United States.

(b) In designating uses of a water body and the appropriate criteria for those uses, the State shall take into consideration the water quality standards of downstream waters and shall ensure that its water quality standards provide for the attainment and maintenance of the water quality standards of downstream waters.

(c) States may adopt sub-categories of a use and set the appropriate criteria to reflect varying needs of such sub-categories of uses, for instance, to differentiate between cold water and warm water fisheries.

(d) At a minimum, uses are deemed attainable if they can be achieved by the imposition of effluent limits required under sections 301(b) and 306 of the Act and cost-effective and reasonable best management practices for nonpoint source control.

(e) Prior to adding or removing any use, or establishing sub-categories of a use, the State shall provide notice and an opportunity for a public hearing under § 131.20(b) of this regulation.

(f) States may adopt seasonal uses as an alternative to reclassifying a water body or segment thereof to uses requiring less stringent water quality criteria. If seasonal uses are adopted, water quality criteria should be adjusted to reflect the seasonal uses, however, such criteria shall not preclude the attainment and maintenance of a more protective use in another season.

(g) States may remove a designated use which is *not* an existing use, as defined in § 131.3, or establish sub-categories of a use if the State can demonstrate that attaining the designated use is not feasible because:

(1) Naturally occurring pollutant concentrations prevent the attainment of the use; or

(2) Natural, ephemeral, intermittent or low flow conditions or water levels prevent the attainment of the use, unless these conditions may be compensated for by the discharge of sufficient volume of effluent discharges without violating State water conservation requirements to enable uses to be met; or

(3) Human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place; or

(4) Dams, diversions or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use; or

(5) Physical conditions related to the natural features of the water body, such as the lack of a proper substrate, cover, flow, depth, pools, riffles, and the like, unrelated to water quality, preclude attainment of aquatic life protection uses; or

(6) Controls more stringent than those required by sections 301(b) and 306 of the Act would result in substantial and widespread economic and social impact.

(h) States may not remove designated uses if:

(1) They are existing uses, as defined in § 131.3, unless a use requiring more stringent criteria is added; or

(2) Such uses will be attained by implementing effluent limits required under sections 301(b) and 306 of the Act and by implementing cost-effective and reasonable best management practices for nonpoint source control.

(i) Where existing water quality standards specify designated uses less than those which are presently being attained, the State shall revise its standards to reflect the uses actually being attained.

(j) A State must conduct a use attainability analysis as described in § 131.3(g) whenever:

(1) The State designates or has designated uses that do not include the uses specified in section 101(a)(2) of the Act, or

(2) The State wishes to remove a designated use that is specified in section 101(a)(2) of the Act or to adopt subcategories of uses specified in section 101(a)(2) of the Act which require less stringent criteria.

(k) A State is not required to conduct a use attainability analysis under this regulation whenever designating uses which include those specified in section 101(a)(2) of the Act.

§ 131.11 Criteria.

(a) *Inclusion of pollutants:* (1) States must adopt those water quality criteria that protect the designated use.