Wisconsin 2014 Consolidated Assessment and Listing Methodology (WisCALM)

Clean Water Act Section 305(b), 314, and 303(d) Integrated Reporting

Wisconsin Department of Natural Resources
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WISCONSIN
DEPT. OF NATURAL RESOURCES

BUREAU OF WATER QUALITY
PROGRAM GUIDANCE

Wisconsin 2014 Consolidated Assessment and Listing Methodology (WisCALM) for Clean Water Act Section 305(b), 314, and 303(d) Integrated Reporting

September 2013

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Background

Over 15,000 lakes and 84,000 miles of streams and rivers in Wisconsin are managed to ensure that their water quality condition meets state and federal standards. Water quality standards (WQS) are the foundation of Wisconsin's water quality management program and serve to define goals for a waterbody by designating its uses, setting criteria to protect those uses, and establishing provisions to protect water quality from pollutants.

Waters are monitored to collect water quality data to determine, or *assess*, its current status or condition. Water quality monitoring results and assessment data are stored in state and federal databases and the majority of data are available online to agencies and the public. *General assessments* are known as "305(b) assessments" in the Federal Clean Water Act. Waters with available data are reviewed by Wisconsin Department of Natural Resources (WDNR) biologists and placed in one of four categories: excellent, good, fair and poor.

Specific assessments are conducted to determine if a waterbody is "impaired" or not meeting WQS. Waters that do not meet WQS are placed on Wisconsin's Impaired Waters List—also known as the 303(d) list—under Section 303(d) of the Clean Water Act. Wisconsin is required to submit list updates every 2 years to the United States Environmental Protection Agency (EPA) for approval. WDNR has submitted Impaired Waters Lists, as required[1], every other year since 1996.

Water quality assessments aid Department staff in determining management actions that are needed to meet WQS, including anti-degradation, or maintenance, of existing water quality condition, as well as restoration of impaired waters.

Each state must document the methodology used to assess waters, including how the state makes decisions to add or delete waters from the existing Impaired Waters List. Waters may be removed from the list (delisted) when water quality data identifies that the designated use has been restored (i.e., the water is meeting WQS). The methodology for conducting general and specific assessments is outlined, and updated for 2014, in this Wisconsin Consolidated Assessment and Listing Methodology (WisCALM) guidance document.

[1] EPA did not require and WDNR did not submit an Impaired Waters List in FFY 2000.

1.0 Water Quality Standards: Three Elements

Wisconsin's assessment process begins with water quality standards (WQS). WDNR is authorized to establish WQS that are consistent with the Federal Clean Water Act (Public Law 92-500) through Chapter 281 of the Wisconsin Statutes. These WQS are explained in detail in chs. NR 102, 103, 104, 105, and 207 of the Wisconsin Administrative Code (Wis. Adm. Code).

The WQS described in the Wis. Adm. Code rely on three elements to collectively meet the goal of protecting and enhancing the state's surface waters:

- *Use designations*, which define the goals for a waterbody by designating its uses,
- *Water quality criteria*, which are set to protect the water body's designated uses, and
- *Anti-degradation provisions* to protect water quality from declining.

Waters not meeting one or more of these water quality elements are to be included on the Impaired Waters List.

Designated Uses

Designated uses are goals or intended uses for surface waterbodies in Wisconsin which are classified into the categories of: Fish and Aquatic Life, Recreation, Public Health and Welfare, and Wildlife. The following designated uses are described in ch. NR 102, Wis. Adm. Code:

- *Fish and Aquatic Life:* All surface waters are considered appropriate for the protection of fish and other aquatic life. Surface waters vary naturally with respect to factors like temperature, flow, habitat, and water chemistry. This variation allows different types of fish and aquatic life communities to be supported. This category has subcategories as described below.
- *Recreational Use:* All surface waters are considered appropriate for recreational use unless a sanitary survey has been completed to show that humans are unlikely to participate in activities requiring full body immersion.
- *Public Health and Welfare:* All surface waters are considered appropriate to protect for incidental contact and ingestion by humans. All waters of the Great Lakes as well as a small number of inland water bodies are also identified as public water supplies and have associated water quality criteria to account for human consumption[2].
- *Wildlife:* All surface waters are considered appropriate for the protection of wildlife that relies directly on the water to exist or rely on it to provide food for existence.

Use Designations for Fish and Aquatic Life (FAL) are separated into the following sub-categories: Coldwater (Cold), Warmwater Sport Fish (WWSF), Warmwater Forage Fish (WWFF), Limited Forage Fish (LFF) and Limited Aquatic Life (LAL). More detail on these subcategories is located in the Streams and River Classification chapter of this report.

Water Quality Criteria – Numeric and Narrative

Each designated use has its own set of water quality criteria, either numeric or narrative requirements that must be met to protect the intended use. Some of these requirements relate to the amount of the physical (e.g., water temperature) or chemical (e.g., ammonia concentrations) conditions that must be met to avoid causing harm. Wisconsin's water quality criteria may be either numeric (quantitative) or narrative

[2] Distinct water quality criteria are specified for public water supply and non-public water supply waters. Wisconsin does not currently have a formal "Drinking Water" use designation in its standards. Establishment of a "Drinking Water" use designation may be considered as part of a future standards change. If so, specific drinking water use assessment procedures will be included in future updates to the WisCALM document.

(qualitative) and are authorized by state statutes and enumerated in chs. NR 102, 104, and 105, Wis. Adm. Code.

Numeric criteria: Numeric criteria are quantitative and are expressed as a particular concentration of a substance or an acceptable range for a substance. For example, the pH value shall be from 6-9 standard units. Numeric surface water quality criteria have been established for conventional parameters (e.g., DO, pH, and temperature), toxics (e.g., metals, organics, and ammonia), and pathogens (e.g., *E. coli* and fecal coliform bacteria). These numeric criteria are established for each designated use.

Narrative criteria: All waterbodies must meet a set of narrative criteria which qualitatively describe the conditions that should be achieved. A narrative water quality criterion is a statement that prohibits unacceptable conditions in or upon the water, such as floating solids, scum, or nuisance algae blooms that interfere with public rights. These standards protect surface waters and aquatic biota from eutrophication, algae blooms, and turbidity, among other things. The association between a narrative criterion and a waterbody's designated use is less well defined than it is for numeric criteria; however, most narrative standards protect aesthetic or aquatic life designated uses. Wisconsin's narrative criteria are found in s. NR 102.04(1), Wis. Adm. Code.

Anti-degradation
Wisconsin's anti-degradation policy is intended to maintain and protect existing uses and high quality waters. This part of a waterbody quality standard is intended to prevent water quality from sliding backwards and becoming poorer without cause, especially when reasonable control measures are available. The anti-degradation policy in Wisconsin is stated in s. NR 102.05(1) of the Wis. Adm. Code:

> "No waters of the state shall be lowered in quality unless it has been affirmatively demonstrated to WDNR that such a change is justified as a result of necessary economic and social development, provided that no new or increased effluent interferes with or becomes injurious to any assigned uses made of or presently possible in such waters."

One component of Wisconsin's anti-degradation policy is the designation of Outstanding Resource Waters (ORW) and Exceptional Resource Waters (ERW). These are surface waters which provide outstanding recreational opportunities, support valuable fisheries and wildlife habitat, have good water quality, and are not significantly impacted by human activities. ORWs typically do not have any dischargers, while ERW designation offers limited exceptions for dischargers if human health would otherwise be compromised (e.g., expansion of wastewater treatment facilities to protect public health).

Inherent in the assessment and impaired waters listing process is the application of anti-degradation provisions. Anti-degradation is an important aspect of pollution control because preventing deterioration of surface waters is less costly to society than attempting to restore waters once they have become degraded.

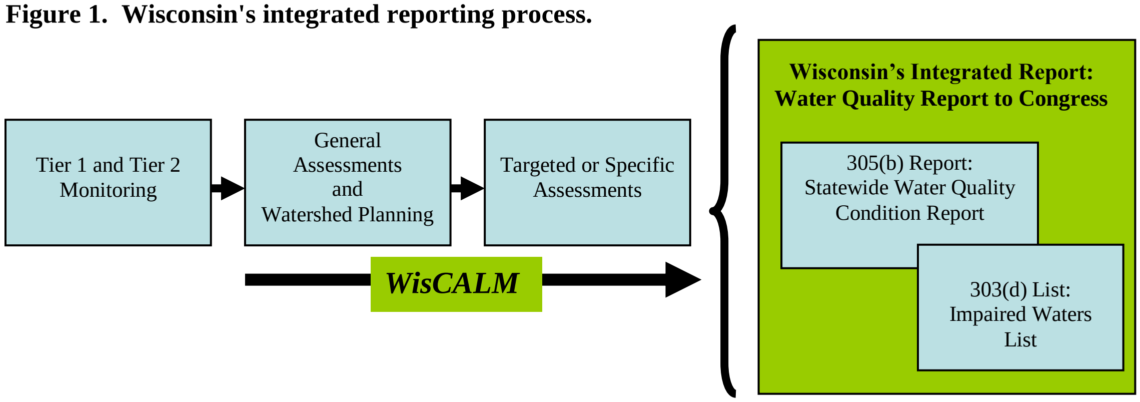
2.0 Wisconsin's Monitoring Program and Data Management

Wisconsin DNR's Water Division Monitoring Strategy is available on WDNR's website at: http://dnr.wi.gov/topic/SurfaceWater/monitoring.html

2.1 Three Tiers of Monitoring

WDNR's Surface Water Monitoring Strategy[3] directs monitoring efforts in a manner that efficiently addresses the wide variety of information needs, while providing adequate depth of surface water knowledge to support decision making. This monitoring strategy employs a three-tiered approach to information gathering to ensure that the status of Wisconsin's water resources can be determined in a comprehensive manner without depleting the capacity to conduct in-depth analyses and problem-solving where needed. The first two tiers of monitoring allow the state to assess waters and place evaluated waters into condition categories (excellent, good, fair and poor) as reflected in the Integrated Report, including the Impaired Waters List (Figure 1).

Figure 1. Wisconsin's integrated reporting process.



Three tiers of monitoring are incorporated into the Integrated Reporting Process:

Tier 1 – Statewide Baseline Monitoring: *Establishing Trends*
Under Tier 1 of the monitoring strategy, staff and partners collect baseline condition information to help satisfy Water Division information needs on a broad spatial scale. Tier 1or baseline monitoring helps obtain broad-scale, statewide assessments of Wisconsin's waters. This procedure is helpful when water resources are too numerous to evaluate individually. Wisconsin's over 84,000 stream miles, for example, call for this dispersed sampling effort which provides, through inference, technically rigorous and credible 'snapshot' of statewide water conditions. Baseline monitoring work provides core information for the state's Clean Water Act general assessment work (305(b)); however, the terms "Tier 1 monitoring" and "General Assessments" are not synonymous. A general assessment is simply reviewing existing data and consistently applying key parameters and minimum results to waters within a given area. This broad scale analysis identifies waters needing further evaluation or "specific assessments."

Under the tiered approach, metrics collected through Tier 1 monitoring include:

<u>*Lakes*</u>

Trophic Status Index (TSI)*
Aquatic Macrophyte Community Index (AMCI) *

[3] WDNR Water Division Monitoring Strategy, Nov. 2008. Wisconsin Department of Natural Resources, Madison, WI.

Contaminants in fish tissue—mercury and PCBs*
Pathogen indicators *
Game fish population dynamics

Streams and Rivers

Macroinvertebrate samples*
Fish assemblage characteristics*
Water chemistry*
Contaminants in fish tissue—mercury and PCBs *
Pathogen indicators*
Gamefish, Endangered, & Threatened species surveys
Habitat assessment

** Metrics used in the general assessment steps are described in Chapters 4.2 and 5.2 of this report.*

Tier 2 – Targeted Evaluation Monitoring: *Site-specific Monitoring*
Sites on waterbodies identified under Tier 1 as potentially being impaired are prioritized based on professional judgment and available resources and may be monitored more intensively under Tier 2 monitoring. Tier 2 is often used to verify whether waterbodies should be placed on the Impaired Waters List and to develop comprehensive water quality management plans or Total Maximum Daily Loads (TMDLs). Under this tier, confirmation of the impairment is made, along with documentation of the pollutant and possible cause(s). For instance, Tier 2 monitoring might focus on resurveying 'flagged' Tier 1 sites and expanding monitoring along the waterbody to determine whether a problem really exists, and the extent of the problem. Or, Tier 2 monitoring might be used to determine what the cause of the impairment is. Thus, it is a more comprehensive evaluation of individual waterbodies, often requiring cross-program collaboration. Tier 2 monitoring may also provide baseline data to determine how well a waterbody responds to management, as evaluated under Tier 3.

Tier 3 – Management Effectiveness and Compliance Monitoring: *Determining effectiveness of management practices and permit conditions*
Tier 3 monitoring evaluates management practices that have been implemented through TMDL implementation or a nonpoint source nine key elements plan. Tier 2 monitoring may also provide information for evaluating permit compliance and effectiveness. Effluent monitoring helps WDNR determine whether permitted entities are meeting their permit conditions and state regulations, and to assess the health of waters receiving effluent. Monitoring of public drinking water wells is also carried out under Tier 3 to ensure that surface and groundwater meet federal public health standards for contaminants in drinking water. Effectiveness of water-specific management actions is determined using core indicators from the more intensive sampling designs under Tier 2 that are specific to the problem being addressed. The chosen indicators are compared before and after management actions are implemented.

2.2 Use of Monitoring Data from Other Sources

In addition to Department-generated data, WDNR biennially seeks information from partners and the public to use in its assessment of waterbodies. Partners include: the U.S. Geological Survey, EPA, U.S. Fish and Wildlife Service, other state agencies, universities, regional planning commissions and major municipal sewerage districts. Guidance is provided on how to submit third party data on the WDNR website. GovDelivery, a web-based service used by WDNR, was also used to solicit data from citizens. This service offers the public real-time updates on topics of interest via email or text messages, and is also used to provide information regarding the Integrated Reporting Process and Wisconsin's Impaired Waters Program.

As datasets are submitted, WDNR reviews the data and the procedures used to collect and analyze the data. WDNR will review information provided by any individual or group at any time; however, the data used for listing purposes must have been obtained using documented quality assurance procedures that meet WDNR procedures. WDNR has an internal website that outlines our State Quality Management Plan. Data submitters outside of WDNR are referred to EPA's site for questions on quality assurance project plans at http://www.epa.gov/QUALITY/qapps.html.

Agencies and individuals submitting data for assessment purposes must: meet minimum data requirements, demonstrate that sample collection occurred at appropriate sites, during appropriate periods, and use certified laboratories for sample analysis. If the quality assurance procedures are not adequate, staff may use this data to initiate further investigations by Department staff. If quality assurance procedures are adequate, WDNR may use this data to assess the water for possible impairment listing.

WDNR may assist outside groups in the design and implementation of data quality procedures necessary for data to be used for assessments. Department staff will consult with EPA water quality criteria guidance, state WQS, and use professional judgment to interpret the results of field sampling to determine whether or not WQS are achieved. Groups outside of WDNR who regularly collect and submit data to WDNR may work with staff at Central Office to upload data into the SWIMS database to be considered as part of our evaluation and assessment process.

WDNR also supports a Citizen Based Monitoring Program for rivers, streams and lakes. As stated in the WDNR's Water Resources Monitoring Strategy for Wisconsin, “If citizens follow defined methodology and quality assurance procedures, their data will be stored in a Department database and used in the same manner as any Department-collected data for status and trends monitoring defined in the Strategy.” Citizen data are currently used for general water quality assessments, including broad-scale statewide assessments. If these data indicate a potential water quality problem at a specific site, additional data may be collected by Department staff to verify the extent of the problem and determine if a waterbody should be placed on the Impaired Waters List.

2.3 Quality Assurance and Laboratory Analysis

Information used for assessments must be consistent with the WDNR Quality Management Plan or have been obtained using comparable quality assurance procedures. For all Tier 1 (baseline) monitoring supporting general and statewide assessments, quality assurance measures are described within each applicable chapter of the *Wisconsin DNR Water Division Monitoring Strategy*. WDNR uses only certified laboratories sample analysis, primarily the State Lab of Hygiene and the University of Wisconsin Stevens Point Aquatic Entomology Laboratory. For targeted, or special, monitoring studies which are frequently used to discern impairment prior to listing a waterbody, quality assurance protocols, such as field blanks, duplicates or spikes, are incorporated as funds allow.

2.4 Data Management

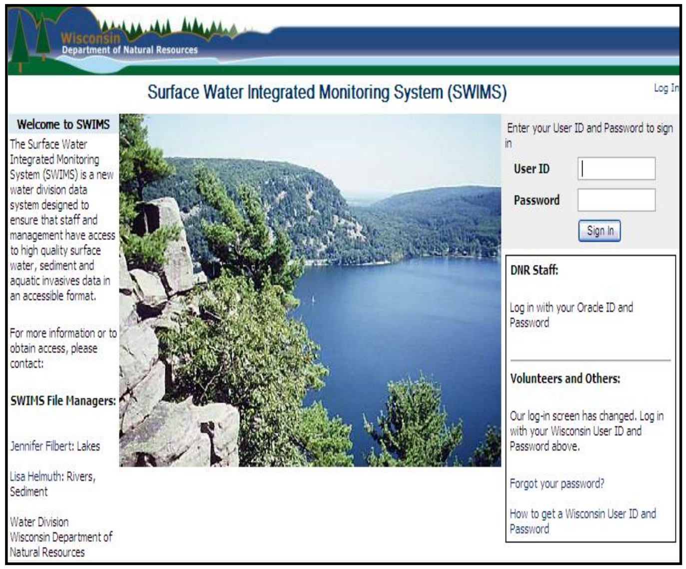
Well organized and readily accessible data is fundamental to a smooth functioning, scientifically grounded water quality monitoring and assessment program. The WDNR has invested many resources into building and maintaining monitoring and assessment databases.

Monitoring Data –SWIMS

The Surface Water Integrated Monitoring System (SWIMS) (Figure 2) is a WDNR information system that holds chemistry (water, sediment), physical (flow), and biological (macroinvertebrate, aquatic invasive) data.

SWIMS is the state's repository for water and sediment monitoring data collected for Clean Water Act work and is the source of data sharing through the federal Water Quality Exchange Network, which is an online federal repository for all states' water monitoring data. WDNR Fisheries and Water Quality Biologists use the system to document monitoring stations for both Water Quality and Fisheries Program datasets, providing a gateway to fisheries management datasets housed at the U.S. Geological Survey.

Figure 2. SWIMS database sign in screen.



The SWIMS database supports Citizen Based Stream Monitoring (CBSM) Level 2 Program volunteers. Level 2 volunteers come into the program with previous water monitoring experience, most volunteers having participated in the CBSM Level 1 Program (Water Action Volunteers or WAV Program). The Level 2 training focuses on the proper use of WDNR field methods and specialized equipment, such as transparency tubes, DO and pH meters. The Level 2 Program Coordinator trains volunteers to properly calibrate the instruments, use and store the equipment, record the data, etc. Volunteers chose monitoring locations on nearby streams with input from WDNR staff. The data collected by Level 2 volunteers are entered into the SWIMS database and quality assured by WDNR staff. SWIMS also supports the Citizen Lake Monitoring Network (CLMN) datasets, which are collected by citizen volunteers and used directly for lake general assessment work.

Assessment Data -- WATERS

The Water Assessment, Tracking and Electronic Reporting System (WATERS) is a data system that includes the following water program items:

- Water Division Objectives, Goals, Performance Measures, and Success Stories
- Clean Water Act Use Designations and Classifications (chs. NR 102 and 104, Wis. Adm. Code)
- Outstanding and Exceptional Resource Waters Designations (ch. NR 102, Wis. Adm. Code)
- Clean Water Act assessment data, including decisions about whether a waterbody is meeting its designated use or is considered "impaired"
- impaired waters tracking information, including the methodology used for listing, the status of the TMDL creation, and restoration implementation work
- Fisheries Trout Classifications (s. NR1.02(7), Wis. Adm. Code)
- Watershed planning recommendations, decisions, and related documents

2.5 Data Requirements

By establishing data requirements, WDNR staff collect representative data as efficiently as possible with limited staff and fiscal resources and use those data in a manner that minimizes the chance of incorrectly characterizing the attainment status of a particular water. Extremely large datasets are neither available nor necessary for many water bodies in the state. Minimum data requirements have been established for indicators including:

- **Period of Record:** Generally, data from the most recent 10-year period may be considered when assessing waters to ensure that the data are representative of a wide range of factors that affect water quality (i.e., weather, flow) [4]. If staff determine that older data within the 10-year period are no longer representative of recent conditions, the period may be shortened to the most recent 5 years. To make such a determination department staff will consider whether significant changes in the watershed have occurred, such as changes in land use, nonpoint source controls, or the amount of pollutants discharged from point sources.

- **Sampling Period:** The WisCALM guidance document identifies the appropriate sampling period for each parameter and waterbody type. The determination of appropriate sampling period is based on seasonal variability in pollutant levels and corresponding ecological responses. Data from two sampling seasons will be needed for some assessments to account for sampling error or annual variation.

- **Representative Data:**

    - **Sampling Protocol**: Individual data points must have been collected according to parameter-specific protocols. Prescheduled sampling designs are often used for 305(b)/303(d)-related monitoring in order to randomly capture the range of conditions. In these cases, targeted samples that are collected for other purposes (e.g. monitoring targeted during runoff events) should not be incorporated into the 305(b)/303(d) assessment datasets. In other cases, weather and hydrologic conditions must match intended conditions specified in the sampling protocols. For example, biological samples should be collected during base flow, not following a runoff or scouring flow event, to ensure the sample is representative of normal conditions.

    - **Extreme Weather Years:** Chemical and biological parameters are likely to be affected by extreme weather conditions. If a prescribed sampling schedule falls during an extreme weather year, exhibiting unusual average air temperature, precipitation, stream flow or water levels, a determination should be made as to whether that year was an extreme weather year that resulted in unrepresentative conditions. As a very general guideline, an extreme weather year may be defined as a year where precipitation, flow, stage/elevation, and/or temperature are above the 90th or below the 10th percentile of the annual averages within the period of record. Staff may use a combination of the following sources to document their determination of whether data were collected from a particular waterbody during an extreme weather year:
        - Climate data from nearest regional weather station(s)
        - Regional stream stage/flow gage(s)
        - Indices of drought severity (e.g., Palmer Drought Severity Index, U.S. Drought Monitor)

[4] Total phosphorus and biological data (chlorophyll, macroinvertebrates and fish) from the most recent 5-year period are used to make impairment decisions. However, if insufficient data are available from the most recent 5-year period, data collected within the past ten years may be used.

If it is determined that a year was an extreme weather year resulting in unrepresentative conditions, that year's data points should not be excluded, but rather should be supplemented with data from an additional year of monitoring. In this case, combined data from a minimum of two years should be used for assessments to account for variability between years. Gaps in assessment datasets left when samples are determined to be unrepresentative should be filled by either collecting additional data or considering data from outside the standard period of record.

Best professional judgment may be used to determine whether data were collected from an extreme weather year and are considered unrepresentative of normal conditions. For instance, a region may be experiencing drought, but stream flow may not be impacted significantly for those streams that are dominated by groundwater flows.

- **"Evaluated" Information:** Information that is not considered representative of current conditions or was not collected according to WDNR's Quality Management Plan cannot be used in preparation of the Impaired Waters List. WDNR classifies these types of data as "evaluated" information, which may include:
  - Information provided by groups, other agencies or individuals where collection methods are not documented and thus the data quality cannot be assured
  - Projected surface water conditions based on changes in land use with no corresponding in-water data (i.e., desktop analyses or models)
  - Visual observations that are not part of a structured evaluation
  - Anecdotal reports

  Though not used directly to update the impaired waters list, "evaluated" data may potentially be used to identify areas where further monitoring may be needed for future assessment cycles.

- **Sample Type:** The indicator being evaluated will dictate what type of samples should be used for an assessment decision. In some cases, samples may be collected as instantaneous measurements vs. continuous measurements. In other cases, the choice may be between a grab sample and a composite sample. In either case, the selection of the values should result in using the most representative data available.

- **Sample Size:** This document outlines sample sizes that appropriately and efficiently represent existing and relevant conditions. Sample size requirements differ by water body type and parameter. The number of samples required is commensurate with the inherent sampling error and annual variation of the parameter measured. Available representative data should be reviewed to ensure that the minimum data requirements are met. However, a waterbody may be listed as impaired despite minimum sample size not being achieved if overwhelming evidence of impairment exists (see Ch. 7, Professional Judgment).

3.0 The Assessment Process: An Overview

3.1 General Condition Assessment

Data collected under WDNR's tiered monitoring strategy are used to identify where a specific waterbody falls on a continuum of water quality condition, which is the first step in assessing whether a waterbody is attaining its assigned designated uses.

WDNR uses four levels of condition to represent waters' placement in the overall water quality continuum (Figure 3). Waters assigned the condition category of *excellent* are considered to be attaining applicable WQS and ***fully** supporting* their assessed designated uses. Waters assigned the condition category of *good or fair* are *also* considered to be attaining applicable WQS and *supporting* their assessed designated uses. Waters assigned the *poor* condition category *may not be attaining* WQS or assessed designated use(s). Waters determined to be in poor condition based on Tier 1 monitoring data are further evaluated and may be selected for additional (Tier 2) monitoring or, if the limited dataset includes overwhelming evidence of impairment (e.g. large magnitude of exceedance), considered "impaired" and added to Wisconsin's Impaired Waters List.

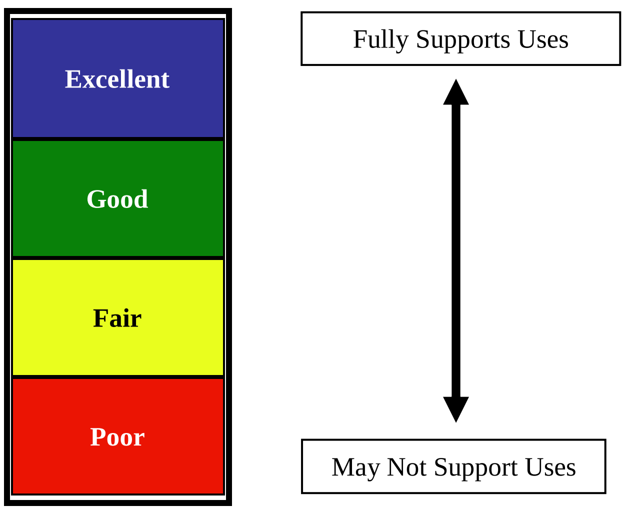


Figure 3. General water condition continuum.

3.2 Impairment Assessment

The assessment of whether a waterbody is meeting designated uses requires comparison to applicable water quality criteria, or, when numeric criteria do not exist, a well-defined reference condition or listing thresholds as a benchmark for comparison to narrative standards.

This section briefly outlines the concepts of indicators and associated thresholds to measure attainment status of Wisconsin lakes, rivers, and streams. For purposes of this guidance, the term "indicator" is used to describe the various measures of water quality, including those that represent physical, chemical, biological, habitat, and toxicity data. The term "threshold" is used to when referring to the numeric value or narrative description that distinguishes attainment of the WQS versus values that indicate impairment. In the simplest sense, a waterbody is defined as "impaired" when it is not meeting WQS, including its assigned designated uses.

Key Indicators for Assessments

Detailed assessments are tailored to the specific characteristics of a waterbody. Some assessments will focus upon one key indicator only, whereas others use multiple indicators. Furthermore, a stepwise process of indicator selection may be employed. For example, for assessment of total phosphorus impacts in cases of moderate enrichment, available biological information will be used to determine fish and aquatic life use impairment and place the water in the proper reporting category. However, if phosphorus levels are exceedingly high, biological indicator data is not needed to determine impairment (i.e., the biological impairment is assumed). Assessment indicators are sub-divided into the following categories:

- Conventional physical-chemical
- Toxicity
- Biological

Other indicators, including habitat metrics, are also used for certain assessments.

Impairment Thresholds

Impairment thresholds are applied to determine whether waterbodies should be placed on the Impaired Waters List. These thresholds are usually expressed as ambient water concentrations of various substances based on numeric water quality criteria included in chs. NR 102-105, Wis. Adm. Code, WDNR technical documents, and federal guidance. In some cases, qualitative thresholds based upon narrative standards may be used to make impairment decisions. In those cases, a thoroughly documented analysis of the contextual information should be used in conjunction with professional judgment to collectively support a decision. Impairment thresholds outlined in WisCALM guidance must be in line with the intent of the water quality criteria in code. In some cases, WisCALM lists impairment thresholds for parameters for which water quality criteria have not been promulgated (e.g., macroinvertebrate and fish indices of biotic integrity and chlorophyll concentration) that may also be used as guidance for impairment listing decisions.

For some assessments methods, a single criterion or threshold may not be applicable across all the different waterbody types. For assessments of waters against the statewide total phosphorus criteria, for example, an initial waterbody classification analysis is required to ensure the assessment process applies the correct criteria. For other assessment methods, the WDNR applies the same water quality criterion or threshold across all resource types. An example is the use of the same fish tissue mercury concentration for all our lakes and rivers in the assessment of Fish Consumption Advisories as part of the Public Health and Welfare Use (Chapter 6.1).

Exceedance Frequency

In the context of numeric water quality criteria, exceedance frequency refers to the number of times a criterion may be exceeded over a period of time before the water is no longer attaining the criterion and is considered impaired. Allowable exceedance frequencies for criteria contained in Wis. Adm. Code, are outlined in this WisCALM document. In addition, allowable exceedance frequencies for some water quality or biological thresholds that are not included in Wis. Adm. Code are provided in the Lakes and Rivers/Streams chapters.

4.0 Lake Classification and Assessment Methods

4.1 Lake Classification

WDNR classifies or groups similar lake types based upon physical data. Specifically, lake size, stratification characteristics, hydrology and watershed size are identified as the primary influences on a lake and, to a large degree, these characteristics determine the natural biological communities each lake type supports. Using this information, lakes should fall into one of ten natural community types (Table 1).

Table 1. Lake and reservoir natural communities and defining characteristics.

Natural Community	Stratification Status	Hydrology
Lakes/Reservoirs <10 acres – Small	Variable	Any
Lakes/Reservoirs ≥10 acres		
• Shallow Seepage	Mixed	Seepage
• Shallow Headwater	Mixed	Headwater Drainage
• Shallow Lowland	Mixed	Lowland Drainage
• Deep Seepage	Stratified	Seepage
• Deep Headwater	Stratified	Headwater Drainage
• Deep Lowland	Stratified	Lowland Drainage
Other Classification (any size)		
• Spring Ponds	Variable	Spring Hydrology
• Two-Story Fishery Lakes	Stratified	Any
• Impounded Flowing Waters	Variable	Headwater or Lowland Drainage

The WDNR recognizes that lakes may vary geographically. Spatial data are available for each of the lakes. Regional differences in soils, climate and land use may explain additional variation in the bio-indicator metrics used in the classification of lakes[5]. However, WDNR has determined that lake size, hydrology and depth are more critical factors for initial classification of lakes, and that regional differences are secondary.

For most lakes, the WDNR's automated data packages determine which natural community and which impairment thresholds are appropriate based on the parameters described below. However, if the biologist has information to suggest that a lake's automatically assigned natural community is inaccurate or not representative of the lake, a change to the natural community may be made if reasons for the change are documented. If a Partial Lake Listing is being considered, a different Natural Community

[5] Past Wisconsin studies have used eco-regions to explain landscape variability and EPA has proposed using this framework for assessment (Omernik 1987).

may be assigned to the portion of the lake being considered for a Partial Lake Listing, based on site characteristics that are significantly different from those in the rest of the lake.

Reservoirs – Reservoirs are classified using the same classification schema as lakes, described below, though biologists may employ multiple sampling stations on reservoirs to provide more representative data. NR 102.06(2)(f) of Wis. Admin. Code defines a reservoir as "a waterbody with a constructed outlet structure intended to impound water and raise the depth of the water by more than two times relative to the conditions prior to construction of the dam, and that has a mean water residence time of 14 days or more under summer mean flow conditions using information collected over or derived for a 30 year period."

Size: Small vs. Large - Lake classification begins by first separating lakes into those 10 acres and greater and those less than 10 acres.

Small Lakes – Lakes less than 10 acres are classified into the Small Lake community. These lakes are uniquely different from communities in larger lakes but there is limited monitoring data available in Wisconsin. Because data for lakes less than 10 acres is so limited, it is difficult to set quality thresholds for assessment. Currently, there are very few thresholds set for water quality, fisheries, or aquatic plants for lakes less than 10 acres[6]. To address these small lakes in the future, Wisconsin may look to emerging wetland assessment tools for guidance.

Large Lakes – Lakes 10 acres or more are classified as Large Lakes. Large Lakes are further subdivided, by stratification status, hydrology, and watershed size, as shown below.

Stratification Status: Shallow (Unstratified or Mixed) vs. Deep (Stratified) – Lakes that are 10 acres or greater may be further characterized by their tendency to mix or stratify thermally. Stratification is an important factor in determining overall lake water quality and availability of suitable habitat for fish and aquatic life. An equation developed by WDNR Researchers (Lathrop and Lillie, 1980) is used by WDNR to identify whether a lake is categorized as Deep (Stratified) or Shallow (Unstratified or Mixed)[7]. Although this model is used to automatically generate lake classifications from the WDNR database, use of field data on depth, area, residence time, and temperature profiles to refine the model-based lake classifications is encouraged.

The Lathrop/Lillie equation is represented by a ratio calculated as follows:

$$\frac{\text{Maximum Depth (meters)} - 0.1}{\text{Log 10 Lake Area (hectares)}}$$

or

$$\frac{\text{Maximum Depth (feet)}*0.3048 - 0.1}{\text{Log 10 (Lake Area (acres)}*0.40469)}$$

Shallow (Unstratified or Mixed) – When using the Lathrop/Lillie Equation, any value less than or equal to 3.8 predicts a mixed lake, which is placed in the Shallow category (Figure 4). Mixed lakes (Figure 5) tend to be shallow, well-oxygenated, and may be

[6] Total Phosphorus criteria apply to lakes of five acres and larger.

[7] WDNR's decision to use the Lillie/Lathrop equation to determine stratification status also examined several other models for predicting lake stratification based on depth and area. These included work by Emmons et al. (1999), the Osgood Index (Osgood 1988), a Minnesota "lake geometry ratio" (Heiskary and Wilson 2005) and a model by WDNR Researchers (Lathrop and Lillie, 1980). The Lathrop/Lillie Equation was selected because it better distinguishes between clearly stratified and mixed lakes.

impacted by sediment re-suspension. In addition, shallow lakes have the potential to support rooted aquatic plants across the entire bottom of the lake (Figure 5).

Figure 5. Illustration of a shallow, mixed lake.

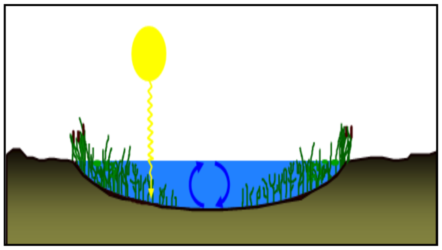
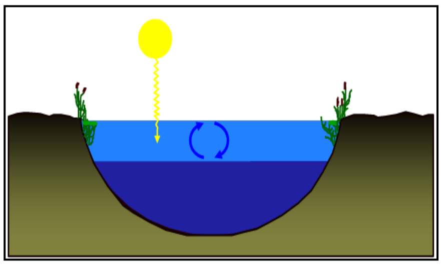
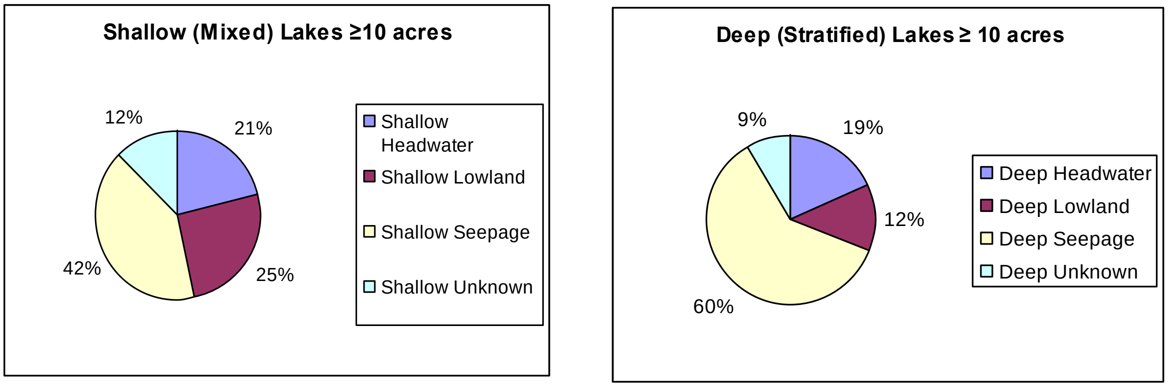


Figure 4. Illustration of a deep, stratified lake.



Deep (Stratified) –When using the Lathrop/Lillie Equation, any value greater than 3.8 predicts a stratified lake, which is placed in the Deep category. Stratified lakes tend to be deep, with a cold water refuge for fish, and the potential for anoxic conditions (without oxygen) in the bottom layer which may release nutrients from sediments into the water column. Aquatic plants are typically confined to shallow (littoral) waters around the perimeter of the lake (Figure 4). Stratified lakes exhibit thermal layering throughout the summer or they undergo intermittent stratification.

Figure 6. Distribution of Shallow and Deep lake types (for lakes greater than 10 acres)



Hydrology and Watershed Size – Lake hydrology is the measure of the relative inflow/outflow of surface water compared to direct precipitation and groundwater inputs. Lake hydrology and lake watershed size are two other critical factors in lake classification. Both Deep and Shallow Lakes are further divided based on hydrology. The terms “seepage” or “drainage” are best used to describe the appropriate hydrologic category for lakes.

Seepage Lakes – A lake with no surface water inflow or outflow is considered a seepage lake (Figure 7). A seepage lake receives water from two sources: primarily from precipitation, both as overland sheet flow to the lake and directly onto the lake and seepage into the lake from groundwater. Seepage lakes tend to have lower nutrient concentrations, due to relatively small catchment areas, and may be poorly buffered against acid deposition.

Drainage Lakes – A lake with surface water inflow/ outflow from a river or stream is classified as a drainage lake (Figure 8). Drainage lakes tend to have more variable water quality and nutrient levels, depending upon the amount of land area drained by the lake’s watershed. For this reason, watershed size also plays a key role in the classification of Drainage Lakes (Emmons, et al, 1999). Drainage lakes are subdivided by watershed size as follows:

- Headwater Drainage Lakes: If the watershed draining to the lake is less than 4 square miles, the lake is classified as a Headwater Drainage Lake.
- Lowland Drainage Lakes: If the watershed draining to the lake is greater than or equal to 4 square miles, the lake is classified as a Lowland Drainage Lake.

Figure 7. Seepage Lake

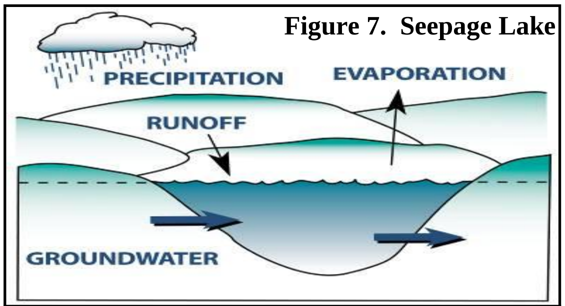
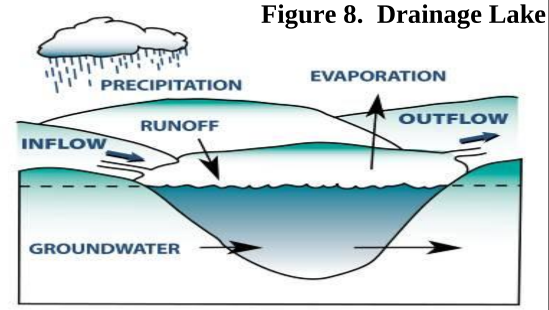


Figure 8. Drainage Lake



Other Classifications (any size) – Three other classes representing unique natural communities are recognized in this classification scheme: Spring Ponds, Two Story Lakes, and Impounded Flowing Waters.

Spring Ponds –Spring ponds typically contain cold surface water and support coldwater fish species and are most often shallow headwater lakes. In order to be included in this category there should be documentation of a current or historical cold water fishery (e.g., stream trout) and evidence of spring hydrology.

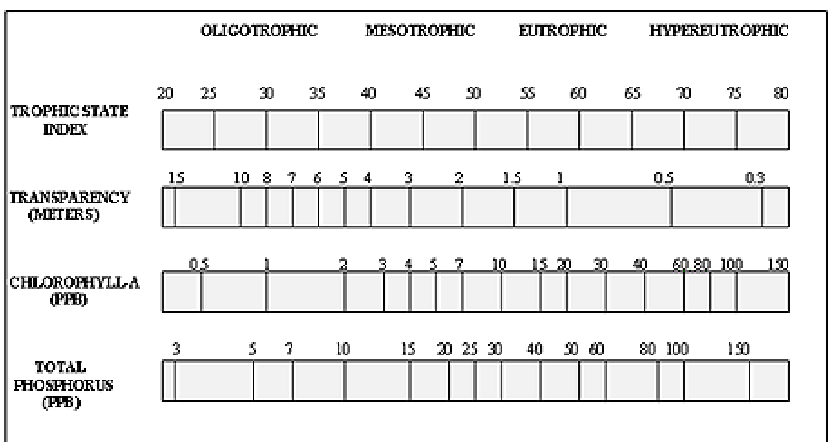
Two Story Fishery Lakes – Two-story fishery lakes are often more than 50 feet deep and are always stratified in the summer. They have the potential for an oxygenated hypolimnion during summer stratification and therefore the potential to support coldwater fish species in the hypolimnion. In order to be included in this category, a lake should meet the definition of "stratified" (Lathrop/Lillie equation value >3.8), be greater than five acres, and support a coldwater fishery. Supporting a coldwater fishery may either be demonstrated through documentation of a current or historical native cold water fishery (e.g., cisco, lake trout), or verification with DNR fisheries biologists that the lake is on a long-term stocking plan for coldwater species, where the individuals have good year-to-year survival.

Impounded Flowing Waters—Rivers or streams that are impounded but do not meet the definition of reservoir above are considered to be "impounded flowing waters." Impounded flowing waters are lotic in nature and should be evaluated using the river and stream criteria that apply to the primary stream or river entering the impounded water. Biological response metrics may also include metrics that are typically used for lakes, such as chlorophyll *a*, as deemed appropriate based on professional judgment.

4.2 Lake General Condition Assessment

The WDNR focuses on in-lake water quality metrics to assess a specific lake's fish and aquatic life designated use. These in-lake parameters correlate strongly with fish and other aquatic life communities (macroinvertebrates, aquatic plants, etc.) within a lake.

Figure 9. Continuum of lake trophic status in relation to Carlson Trophic State Index.



After Moore, L. and K. Thornton, [Ed.]1988. Lake and Reservoir Restoration Guidance Manual. USEPA>EPA 440/5-88-002.

Wisconsin bases its General Condition Assessment for lakes on the Carlson Trophic State Index (TSI). The Carlson TSI is the most commonly used index of lake productivity. It provides separate, but relatively equivalent, TSI calculations based on either chlorophyll *a* concentration (chl *a*, or CHL in the equation below) or Secchi depth (SD, for which Wisconsin also uses satellite clarity data as a surrogate)[8]. Because TSI is a prediction of algal biomass, typically the chl *a* value is a better predictor than Secchi or satellite data. Water clarity as measured by Secchi depth or satellite is a practical measure of algal

[8] Carlson also provides an equation to convert total phosphorus concentration to TSI, but WDNR is not using that equation for purposes of water quality assessments or 303(d) Impaired Waters Listing.

production and water color. Algal production is known to be highly correlated with nutrient levels (especially phosphorus). High levels of nutrients can lead to eutrophication and blue-green algae blooms. This limits the amount of available light to macrophytes and adversely affects other aquatic organisms. Information from each of these parameters is valuable because the interrelationships between them can be used to identify other environmental factors that may influence algal biomass.

TSI values range from low (less than 30), representing very clear, nutrient-poor lakes, to high (greater than 70) for extremely productive, nutrient-rich lakes (Figure 9). Very few lakes in Wisconsin would fall into the category of "very clear, nutrient poor lakes." The cutoff for excellent TSI values would certainly include these lakes (Table 2) but also includes some lakes in the mesotrophic category, based on sediment core data which indicates that some lakes are naturally more productive than others.

Data requirements
TSI is automatically calculated using a programming package (TSI Package) that draws from Department data in SWIMS. The rules used by the TSI Package are described below. These requirements are set to provide enough data to account for the average lake condition during the summer index period (when the lake responds to nutrient inputs and achieves maximum aquatic plant growth) over several years to account for unusual weather (dry, wet, hot, cold). Results from the TSI Package are provided to biologists to use in their assessments. Biologists may use professional judgment in assessing package results.

a) *Seasonal Range and Sampling Frequency.*

- For chl *a* and Secchi data, the TSI Package requires 2 samples per year in each of 3 different years. Samples should be collected between July 15 – September 15.
- For satellite clarity data, at least one satellite inferred clarity reading is required in each of 3 years (3 values minimum). Samples should be collected between July 1 – September 30.

b) *Sampling Depth.* Chlorophyll *a* samples taken from the top 2 meters of the lake will be used to calculate TSI (excluding grab samples collected at 0 m). Samples can be grab samples or integrated samples.

c) *Year Range.* Sampling data are used from within the most recent 5 years (2008-2012).

d) *Sampling and Analytical Methods.* Field collection, preservation and storage should follow procedures outlined in the WDNR Field Procedures Manual and the Citizen Lake Monitoring Manual (http://WDNR.wi.gov/lakes/CLMN/manuals/). Laboratory analysis should follow standard methods (WSLH 1993). Data collected using different protocols may be considered, with limitations, based upon professional evaluation.

Calculations
a) For each year with sufficient data, first all values are converted to TSI using the calculations below (calculate TSI *separately* for chl *a*, Secchi, and satellite data)[9]. (Note: Satellite readings are automatically converted to clarity values (equivalent to Secchi depth) in SWIMS.)

$TSI_{CHL} = 9.81 \ln(CHL) + 30.6$
$TSI_{SD} = 60 - 14.41 \ln(SD)$ (satellite inferred clarity data can also be used in lieu of Secchi data in this equation)

Where:

[9] Although Carlson's Trophic State Index also provides a calculation for TSI based on total phosphorus (TP), Wisconsin does not calculate TSI based on phosphorus for General Condition Assessments. TP concentrations are used to determine whether a waterbody exceeds thresholds for 303(d) listing as a pollutant.

TSI = Trophic Status Index
SD = Secchi depth (meters)
CHL= Chlorophyll *a* concentration (µg/L)
ln = natural log

b) For each year of data, an Annual Average is calculated from the data points within that year (Annual Averages are calculated separately for each parameter).

c) All available Annual Averages from the last 5 years are averaged together, to produce a Multi-year Average (Multi-year Averages are calculated separately for each parameter).

d) The TSI Package automatically prioritizes which TSI Multi-year Average to use in comparison against the General Condition Assessment Thresholds. Historically, there has been a tendency to average the three TSI values, but research suggests that this generally is not a good practice (Carlson and Simpson 1996). Therefore, Wisconsin has instituted a prioritization system for selecting which TSI score to use. When more than one Multi-year Average TSI score is available, whichever TSI score is based on the most direct measure of algal biomass will be used, as follows:

- TSI based on chl *a* will be used if available, since this is the most direct measure of trophic state.
- TSI based on measured Secchi data is the second preference; Secchi depth readings measures clarity as a surrogate for trophic state.
- TSI based on satellite data is the third preference, as it infers water clarity rather than measuring water clarity directly.

e) The final step in the General Assessment is to compare the lake-specific Multi-year Average TSI value to the lake general condition assessment thresholds shown in Table 2. As described previously, the lake condition assessment thresholds establish four categories for each Lake Natural Community: Excellent, Good, Fair, and Poor.

Table 2. Trophic Status Index (TSI) thresholds – general assessment of lake Natural Communities.

Condition Level	Shallow			Deep			
	Headwater	Lowland	Seepage	Headwater	Lowland	Seepage	Two-Story
Excellent	< 53	< 53	< 45	< 48	< 47	< 43	< 43
Good	53 – 61	53 – 61	45 – 57	48 – 55	47 – 54	43 – 52	43 – 47
Fair	62 – 70	62 – 70	58 – 70	56 – 62	55 – 62	53 – 62	48 – 52
Poor	≥ 71	≥ 71	≥ 71	≥ 63	≥ 63	≥ 63	≥ 53

Note: Although TSI thresholds are not yet available for three natural communities: 1) Small Lakes; 2) Spring Ponds; and 3) Impounded Flowing Waters, by default assessments are completed for the most similar natural community for which thresholds are currently available.

Derivation of TSI General Condition Thresholds

Excellent Condition

To establish the excellent range for TSI conditions, WDNR uses excellent or "reference" conditions inferred from total phosphorus (TP) values based upon preserved diatom communities from pre-settlement times found in lake bottom sediment cores.

Sediment cores measure fossilized diatom communities allowing a comparison of historical (pre-settlement) conditions and recent water condition. This allows the comparison of current water clarity measurements to historical conditions with changes represented by the changes in algae conditions over time. Diatoms are a type of algae containing siliceous cell walls that fossilize in lake sediments. Diatom taxa are known to prefer narrow ranges of water quality. Therefore, inferences about historical water condition can be made from fossilized diatom communities at the bottom of the sediment core. These inferred concentrations, when converted to TSI values using the Carlson equations, can be used as reference values. This approach will not work for most reservoirs, impounded flowing waters, or raised wetland lakes since these lakes are artificial and pre-settlement conditions do not exist. WDNR has not yet developed criteria specific to these artificially created waterbodies.

WDNR has sediment core data spanning each of the 6 natural lake community types (Table 3) and derives excellent TSI thresholds from these data (Garrison, unpublished data*). The transition between excellent and good for each natural community is based on the 75th percentile of the TSI values calculated from sediment core bottom inferred phosphorus concentrations.* The bottom sediment core values represent reference lake conditions and using the 75th percentile gives some margin for lakes to have changed since the bottom of the sediment core accumulated (Table 3).

Sediment cores are not available for small lakes or spring ponds and are not appropriate for impounded flowing waters. Since adequate sediment core data from two-story lakes is not available, the 75th percentile value for deep seepage lakes was used for the threshold between excellent and good condition (Table 2). Ideally, sediment core data should be collected whenever monitoring is conducted on two-story lakes.

Table 3. Mean and median inferred TP values calculated from top and bottom segments of sediment cores from 87 Wisconsin lakes (Garrison, unpublished data).

Lake Class	Natural Community	N	Mean TP (µg/L)		Median TP (µg/L)		75^{th} Percentile (µg/L) (Bottom	TSI Threshold
			Top	Bottom	Top	Bottom		
1	Shallow Headwater	17	27	24	26	19	30.3	53
2	Deep Headwater	19	24	18	21	14	20.5	48
3	Shallow Lowland	11	28	25	28	24	30.5	53
4	Deep Lowland	43	25	19	20	15	20.0	47
5	Shallow Seepage	15	17	16	16	14	17.0	45
6	Deep Seepage	29	15	13	12	11	15.3	43

Poor Condition

Setting the threshold for Poor Condition was approached differently for each lake type, as most appropriate for the specific conditions exhibited by those lakes:

> *Shallow Lakes:* The transition between a fair and poor condition for shallow lakes was set at a TSI of 71 (corresponding to TP concentration of 100 µg/L) because this approximates TP concentrations that lead to a switch from aquatic plant dominated to algal dominated ecosystems in shallow lakes (Jeppesen et al. 1990). This represents a major ecosystem change and once it occurs, it is very difficult to restore to the aquatic plant dominated state.
>
> *Deep Lakes:* The fair to poor transition threshold for deep lakes was set using a TSI value known to cause increased frequency of algal blooms, high amounts of blue-green algae and/or hypolimnetic oxygen depletion. A TSI of 63 (corresponding to TP of 60 µg/L) was chosen because it represents the threshold between eutrophic and hyper-eutrophic lakes (Carlson 1977).
>
> *Two-Story Lakes:* TSI values that cause significant hypolimnetic oxygen depletion should be used as the threshold for two-story lakes since this habitat component is critical for maintaining coldwater fisheries. This value will be highly dependent upon the lake's morphometry. Hypolimnetic oxygen demand is largely from the sediment; therefore, the greater the ratio of *sediment area to hypolimnetic water volume* the higher the hypolimnetic oxygen demand. That makes setting this threshold very difficult. A conservative TSI value of 53 (corresponding to a TP

of 30 µg/L) is recommended. Further research on these relationships is needed to derive accurate values for two-story lakes.

Good and Fair Condition
The transition value between the condition of "fair" and "good" for each natural community was selected as a mid-point between the excellent and poor TSI values (Table 2).

4.3 Lake Impairment Assessment: Selecting representative stations and which lakes to evaluate

Not all waters categorized as Poor in the General Condition Assessment should be considered Impaired or warrant 303(d) listing. Whether or not a waterbody should be listed as impaired is dependent on the strength of the data used to make the assessment. To submit a lake for the 303(d) List, it should exceed certain numeric listing thresholds or meet narrative listing criteria. A General Condition Assessment status of "Poor" or "Fair" based on TSI score serves as a flag that TSI values and other parameters such as TP, temperature, DO, and pH should be evaluated against the additional impairment thresholds outlined in Table 5A. In addition, best professional judgment may be needed for certain parameters (such as TSS and turbidity), or unique natural communities (such as two-story lakes or impounded flowing waters) for which there are currently no thresholds or criteria for certain parameters.

It is important to determine the relationship between the impairment and pollutant when placing a waterbody on Wisconsin's Impaired Waters List. There are a number of field-measurements that can be taken to more clearly define the condition of a lake and determine what specific impairments and pollutants may be present. Selecting the correct indicators is an important part of understanding the underlying causes of water quality problems. Collectively, the type of data collected and the frequency of sampling is critical for accurate listing and the development of a successful management strategy. Guidance on how to make attainment decisions for some of the more common pollutants or stressors observed in Wisconsin lakes is provided below.

Station Locations: Selecting representative stations for assessment
Most lakes will use only a single "Deepest Spot" site to characterize the status of the lake. By default, the TP and chl *a* Packages use those sites that are designated as "Deepest Spot" for assessments. If more than one station is designated as "Deepest Spot", the packages will use both. However, biologists can change which stations are selected by the package by using the checkbox in WATERS named "Use for TP/Chlorophyll?". They can select and unselect stations as needed to appropriately characterize the site.

Lakes with multiple stations: Reservoirs, multi-lobed lakes, and very large lakes may not have a Deepest Spot station and/or may need more than one sampling station to accurately characterize the lake's morphology and to assess the lake. In these cases, to determine which stations should be selected to use for assessments, use the following guidelines:

- Typically between two and five stations would be chosen to be representative of lake conditions, depending on the size and character of the lake.
- Select only 'active' stations that have data from within the past ten years.
- If there are stations that seem to be duplicative of the same location, contact SWIMS/WATERS support staff to determine whether those stations should be consolidated.
- For **very large lakes** (Figure 10), select well-spaced stations representative of the entire lake.
- For **reservoirs/flowages** (Figure 11), select stations that are roughly equally spaced along the thalweg (the deepest channel along the river line). Stations in flowing portions near the upstream entry point of the river may be eliminated.
- For **lobed lakes**,
    - o if there are **multiple deepest spots** (Figure 12), select a station for each deep spot.

- if there is **one deepest spot** but it is not representative of the entire lake (Figure 13), select the deep spot as well as other stations to represent the other portions of the lake. It may be more difficult in these situations to determine which stations provide the best representation of the lake.

Once the biologist has selected which stations will be used to assess the lake, the additional stations should be indicated in WATERS. To do this, check the checkbox to the right of each station you wish to select[10]. These stations are then automatically represented in the TP and chl *a* Package results.

For lakes with multiple stations selected, the assessment results for each station will be shown individually.

Note: The maps below are for illustrative purposes only; the stations shown may not be the most representative stations available.

Figure 10. Large Lakes: Select well-spaced stations throughout lake.
Example: Lake Winnebago

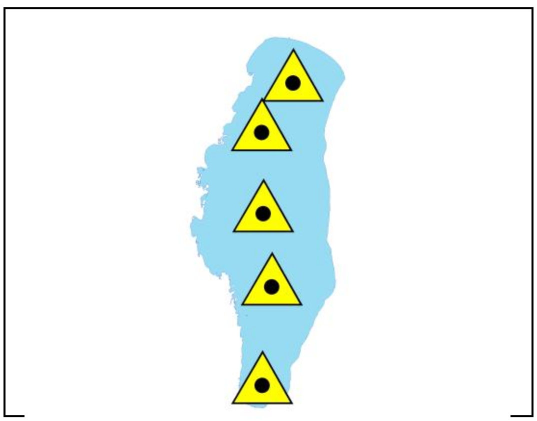


Figure 11. Reservoirs/Flowages: Select stations along the deepest channel.
Example: Lake Petenwell, Juneau County

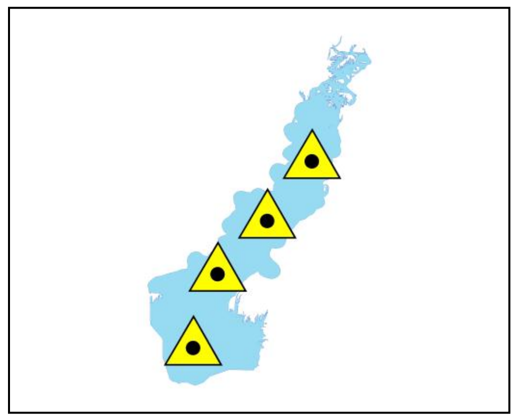


Figure 12. Lobed lakes with multiple deep holes: One station per deep hole.
Example: Two Sisters Lake, Oneida County

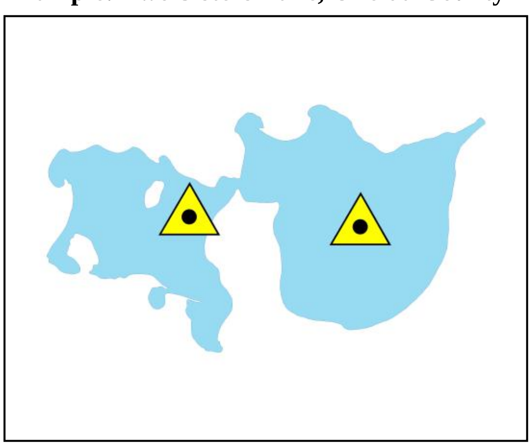
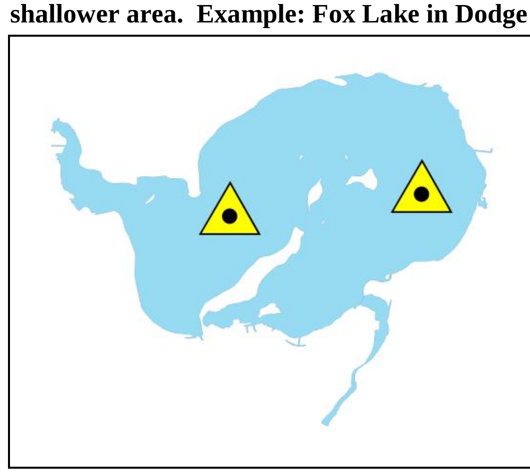


Figure 13. Lobed lakes with one deep hole: Use Deep Hole station and another station representative of shallower area. Example: Fox Lake in Dodge County



[10] Data packages are updated every Friday evening. If new stations are selected, the biologist will need to re-run the packages the following week to incorporate the new information.

Whole Lake vs. Partial Lake Assessment

As a rule, a lake is a mixed system that functions as a single, contiguous unit. Therefore, in the vast majority of situations where there are multiple stations used for assessments, if one station is impaired on the lake, the whole lake would be listed as impaired. However, in cases where a known or suspected localized pollution source is believed to cause impairment in only one portion of a lake (such as an isolated bay or well-defined lobe), biologists may consider assessing and listing that portion as impaired separate from the larger lake.

In cases where Partial Lake Assessments and/or Partial Lake Impairment Listing are warranted, the portion of the lake under consideration should be delineated as a separate Assessment Unit to differentiate it from the larger part of the lake. This is typically warranted when the geography of the lake is such that there is a physical barrier separating most of one portion of the lake from the main portion. In such cases, the partial lake area will typically be assigned its own Natural Community, which may differ from the greater lake.

For Partial-Lake assessments, a sampling station should be added that is representative of the partial-lake area. Such a station should be situated in open water, so that samples are not taken near-shore or in an effluent plume but in ambient lake water within the vicinity of the suspected source of the problem.

Partial Lake Impairment Listings

In cases where a localized pollution source is believed to cause impairment in only one portion of a lake, as evidenced by a station's exceedance of an impairment threshold in only one area of a lake, biologists may consider listing only that portion of the lake as impaired using the appropriate Natural Community threshold. However, if, for instance, one area of a lake is experiencing high algae concentrations due to algae that are being produced throughout the lake but are blown by the wind to a particular area, this would be considered a whole lake problem and partial lake listing would not be appropriate.

4.4 Lake Impairment Assessment: Fish & Aquatic Life (FAL) Uses

Minimum data requirements and calculations for Pollutant and Impairment indicators
For all of the Lake Pollutant and Impairment Indicators, the following guidance on minimum data requirements apply for *Station Location*, *Year Range*, *Sampling and Analytical Methods*, and *Data Quality*. Guidance for frequency, seasonality, sampling depth, and any specific data quality notes are specific to different parameters and are provided under each Pollutant or Impairment Indicator. Some of the more common Pollutants and Impairments are described in the text below; these and others are also documented in Table 5A.

Station Location. See the "Station Location" section in Chapter 4.3.

Sampling and Analytical Methods. Field collection, preservation and storage should follow procedures outlined in the WDNR Field Procedures Manual which is stored in the SWIMS system (http://WDNR.wi.gov/org/water/swims) and the Citizen Lake Monitoring Manual (http://WDNR.wi.gov/lakes/CLMN/manuals/). Laboratory analysis should follow standard methods[11] (WSLH 1993). Data collected using different protocols may be considered, with limitations, based upon professional evaluation of data.

Data Quality. Sample points may be excluded if there are quality control concerns or if the data were collected for specific studies that are not representative of overall lake conditions.

Total Phosphorus (TP) and Chlorophyll *a* (Chl *a*)[12]
Phosphorus is one of Wisconsin's most common pollutants for lakes. In 2010, Wisconsin developed numeric criteria for TP and corresponding protocols for listing waterbodies for TP as a pollutant. Algal biomass, as measured by chlorophyll a concentrations, is one of the most common response metrics to increased phosphorus concentrations. For the purpose of assessing water quality against impairment thresholds, in-lake TP values and chlorophyll a concentrations are calculated using automated programming packages that draw from Department data in SWIMS (these packages are referred to as the TP Package and Chl Package). The rules used by these packages are described below. Results from the packages are provided to biologists to use in their assessments; biologists may use professional judgment in assessing package results.

**New in 2014:*

- *The automated assessment protocols ("Packages") for TP and Chl a have changed significantly for the 2014 assessment cycle, in order to more appropriately assess water quality and to provide more consistency between lake and stream protocols.*
- *<u>Any</u> qualifying data from the period of record in the SWIMS database will be used, and the automated assessment package will provide statistical summary output whether or not the quantity of data points meets the assessment requirements. Including lake datasets that do not meet minimum requirements will allow biologists to review the available data and determine future monitoring needs. However, the automated assessment packages will indicate which stations do or do not meet the minimum data requirements for impairment assessment, and only those that do meet assessment requirements will be used for the automated assessment reporting.*

[11] WSLH (Wisconsin State Laboratory of Hygiene). 1993. Manual of Analytical Methods. Environmental Science Section, Inorganic Chemistry Unit, Wisconsin State Laboratory of Hygiene, Madison, WI.

[12] Heiskary, S, and C. B. Wilson, 2005. Minnesota Lake Water Quality Assessment Report: Developing Nutrient Criteria, Third Edition. Minnesota Pollution Control Agency, September 2005.

TP and Chl have separate thresholds for Recreational (REC) impairments and for Fish & Aquatic Life (FAL) impairments. Therefore, there are four distinct packages that are run to report the needed calculations: TP REC, TP FAL, CHL REC, and CHL FAL. The calculations used are almost identical for TP REC, TP FAL, and CHL FAL. These protocols are described below. The protocols for CHL REC are slightly different and are described in the Chapter 4.5. Once the package results are available, the TP and Chl results are assessed separately and *in combination with one another* to determine whether a lake should be listed as impaired, and if so, in what category. Because algae and aquatic plants are biological metrics that respond to phosphorus, they are used as biological confirmation of impairment related to phosphorus concentrations.

1. Select data to use

Period of record (for both TP & Chl a)
Data from the most recent 10 year period may be used, but data from the most recent 5 years is given preference, as it is more representative of current conditions. See "**Select appropriate year range to use**" (below) for more detail.

Seasonal range and frequency
For official assessment purposes, the goal of the DNR's lake monitoring program will be to have 3 samples per year for both TP and Chl a that meet the data requirements outlined below.

- One sample per month should be taken during the designated sampling season. They should be taken as close as possible to the middle of the month.
- Samples must be spaced at least 15 days apart, to evenly represent the season.
- For TP, the allowable date range is June 1 – Sept. 15, allowing for four monthly samples (June, July, August, Sept.). Only three samples are needed for the calculations, but more samples will be used if available. For Deep (stratified) Lakes, samples from May and/or late September may be manually added if it can be demonstrated that the lake is thermally stratified during that time period.
- For chlorophyll *a*, the target date range is July 15-Sept. 15[13], which should result in one sample for each of July, August, and September. However, if sampling within that window is not possible, data will be accepted if it is collected within one week of the sample season (i.e. July 8-Sept. 22).

Sampling protocols

- *Sampling and analytical methods:* Field collection, preservation and storage should follow procedures outlined in the WDNR Field Procedures Manual which is stored in the SWIMS system (http://WDNR.wi.gov/org/water/swims) and the Citizen Lake Monitoring Manual (http://WDNR.wi.gov/lakes/CLMN/manuals/). Laboratory analysis should follow standard methods[14] (WSLH 1993). Data collected using different protocols may be considered, with limitations, based upon professional evaluation of data.
- *Sampling depth:* Only surface samples taken from the top 2 meters of the lake will be used (excluding grab samples collected at 0 m because these may contain a scum layer). Samples can be grab samples or depth-integrated samples. (If samples were taken from more than one depth

[13] The sampling periods for TP and Chl are not identical. June samples are not used for Chl assessments because many lakes have a clear water phase in June due to food web dynamics. Therefore June samples do not appropriately represent lakes' summer chlorophyll conditions. However, for TP, June samples are included to reflect the range of summer conditions.

[14] WSLH (Wisconsin State Laboratory of Hygiene). 1993. Manual of Analytical Methods. Environmental Science Section, Inorganic Chemistry Unit, Wisconsin State Laboratory of Hygiene, Madison, WI.

within this zone at a single station on a single day, average the samples for that station for that day to produce the station's daily average.)

- *Data quality:* Sample points may be excluded if there are quality control concerns or if the data were collected for specific studies that are not representative of overall lake conditions. See Chapter 2.5 in WisCALM on *Data Requirements*.
- *Units:* Both TP and chlorophyll *a* values should be expressed in ug/L. This is consistent with phosphorus water quality criteria in ch. NR 102, Wis. Adm. Code.

Aggregating samples and determining "qualifying years"

- *Calculate Daily Mean:* Most lakes will have only one sample per day within the correct depth zone (0-2 m or 0-6 ft); in these cases that single sample serves as the daily mean. If there is more than one sample from a single station on a single day from within the correct depth zone, then these samples should be averaged into one, and flagged. Samples with no depth or wrong depth should be excluded.
- *Determine "Qualifying Years"*[15]*:* A "qualifying year" is one that has at least 2 daily means that are in different months of the appropriate date range and that are at least 15 days apart. Whether or not a year is a qualifying year is indicated by the assessment package output.
- *Calculate Monthly Mean:* For all years, regardless of whether they are qualifying years, calculate the monthly mean from the daily means. Most lakes will have only one daily mean per month; in these cases that single value serves as the monthly mean. If more than one daily mean are available for a given month, average them into a monthly mean.

Number of samples required to meet assessment requirements

- For TP, a minimum of 6 monthly means over at least two qualifying years are required.
- For chl *a*, the minimum number of monthly means and years required depends on whether the assessment is being used as a 'biology only' (i.e., standalone) impairment listing for chl *a*, or whether it is being used in conjuction with TP for an impairment listing.
  - For a listing based on biology only (chl *a*) exceedances, a minimum of 6 monthly means over at least two qualifying years are required.
  - For listing based on chl *a* and TP exceedances, a minimum of 3 chl *a* monthly means from at least one qualifying year is required.
- If three monthly means during a year are not available, multiple years may be used to assemble the minimum number of data points.

Select appropriate year range to assess

- All data (that meets requirements for depth/dates/etc.) from the most recent 5 years will be used. If there are enough monthly means within the most recent 5 years to meet minimum data requirements (6 monthly means over at least 2 qualifying years), then only the most recent 5 years will be used.
- If there are not enough monthly means within the most recent 5 years to meet minimum data requirements, then the data package will go back year by year (up to 10 years) to include more months until the minimum data requirement is met, and then stop (i.e. will not use any additional months from the 5-10 year range once minimum data requirement is met).
- If there are not enough months with data from the whole 10 year period to meet the minimum data requirements, the package will still run the formulas and provide statistical summary output

[15] At this stage, biologists may also determine whether any years should be considered "Extreme Weather Years", as described in Chapter 2.5 in WisCALM on *Data Requirements*. If so, and if the biologist feels the extreme weather year resulted in data that would make the assessment result unrepresentative, the biologist may manually check to determine that at least one "normal year" was included in the assessment before making impairment decisions. Gaps in assessment datasets left when samples are determined to be unrepresentative should be filled by either collecting additional data or considering data from outside the standard period of record.

using the months available from that 10 year period, for informational purposes. However, the station will be flagged as not meeting assessment requirements.

2. <u>Compute confidence intervals and exceedance frequencies</u>

The assessment packages run the following calculations on all stations that have any monthly data, regardless of whether they have enough data to meet the minimum data requirements for assessment purposes. However, stations that do not meet the minimum data requirements for an assessment are flagged (see section "**Indicate whether results meet assessment requirements**"). Years that did not have at least 2 monthly means are also flagged.

Along with the automated assessment packages, an Excel spreadsheet template is also available for performing the calculations described below manually. Manual calculations of the statistical values may be required to assess data that is not in the SWIMS database.

Calculate the grand mean[16] and related statistics

Take the average of monthly means across years to calculate each station's grand mean. Use monthly means from the 'appropriate year range' as described above. The grand mean is used for TP REC & FAL, and Chl FAL (not for Chl REC). The list of statistical values needed for this calculation and other values useful for assessment and reporting are:

- Applicable impairment thresholds for the lake type
- Grand Mean
- Min
- Max
- 90% CI –Lower (see formula below)
- 90% CI –Upper (see formula below)
- Standard Deviation
- # of data points used
- Period of Record (the most recent 10 year period, starting with the most recent **even numbered** year)
- Year range used from within the period of record
- Number of years used
- Number of monthly means used

Calculate confidence intervals for TP REC & FAL, & Chl FAL

The following statistical method applies to the Lakes TP package for both FAL and REC. For the Lakes chl a package, it applies for the FAL impairment assessment, but not REC.

The confidence interval (CI) around the mean is:

$$CI = exp\left(\bar{Y} \pm t_{1-\frac{\alpha}{2},N-1}\frac{S}{\sqrt{N}}\right)$$

where $\bar{Y}$ and S are the mean and standard deviation, respectively, of the natural logarithms of the measured values, N is the sample size, α is the desired significance level, and $t_{1-\alpha/2, N-1}$ is the 100(1-α/2) percentile of the t distribution with N - 1 degrees of freedom.

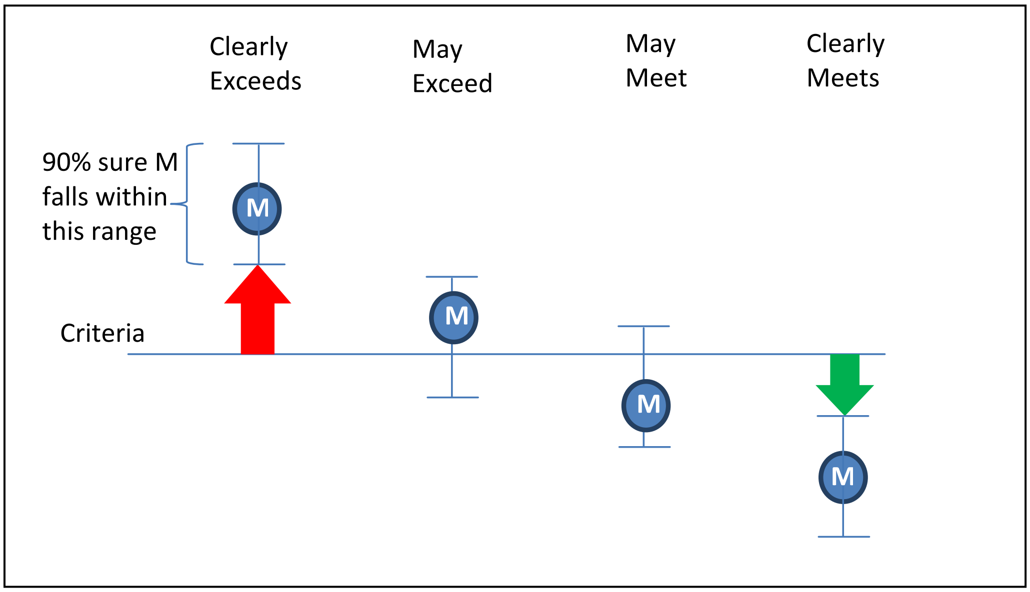
[16] This approach of calculating a grand mean differs from that used in 2012. In the 2012 WisCALM, annual averages were calculated and the number of annual averages that exceeded the applicable thresholds was used to determine impairment. For the 2014 WisCALM, all data points, from multiple years, that meet requirements are combined into a grand mean and its 90% confidence intervals, which are compared against listing thresholds.

3. <u>Compare formula results to the applicable criteria/thresholds</u>

For each of the formula runs above (TP REC & FAL, and CHL FAL), as well as the CHL REC described in the next chapter, compare the resulting Upper and Lower 90% Confidence Intervals to the applicable TP criteria and CHL thresholds for the lake type. The impairment criteria/thresholds for FAL are shown in Table 5 for FAL and in Table 6 for REC.

- If Lower 90% CI > criteria = the lake "Clearly Exceeds" the criteria.
- If Upper 90% CI > criteria = the lake "Clearly Meets" the criteria.
- If Grand Mean > criteria, AND lower CI < criteria, AND Upper CI > criteria = the lake "May Exceed" the criteria.
- If Grand Mean < criteria, AND lower CI < criteria, AND Upper CI > criteria = the lake "May Meet" the criteria.

Regardless of whether the decision was a "Clear" decision, the package will report the decision based upon the data points used to meet the minimum data requirements, rather than including older data that may be less representative[17].



4. <u>Indicate whether the package results meet the assessment requirements</u>

For TP results, indicate the following:

- Did the data meet the minimum data requirements for assessments? (Need at least 6 monthly means, from at least 2 qualifying years.)

For Chl results (both REC & FAL), indicate the following:

- Do the results qualify for an assessment based on TP and chlorophyll? (Need at least 3 monthly means, from at least 1 qualifying year.)
- Do the results qualify for a "biology-only" assessment? (Need at least 6 monthly means from at least 2 qualifying years).

[17] The Integrated Reporting workgroup discussed whether to include more data from earlier years to try to reach a more "Clear" decision, but decided against this. If the lake is trending better or worse over time, it is most appropriate to use the most recent data and recommend future monitoring to reach a more "Clear" decision rather than using older data. However, biologists may incorporate less recent data if they feel it is appropriate to do so.

5. Determine listing categories: Hierarchy of Indicators

Once it has been determined that one or more metrics (TP and/or biological metrics such as chlorophyll or macrophytes) have exceeded an impairment threshold, the department looks at the results of both the TP and biological response indicators in combination to determine which listing category the lake should be placed into. There are several assessment paths that can lead to listing a lake as impaired for TP, chlorophyll *a*, or a combination of both.

- *TP Only—based on "Overwhelming TP exceedance":* If a lake's lower 90% confidence interval exceeds its phosphorus criterion by 1.5 times[18], it is considered to have an 'overwhelming exceedance' of the phosphorus criteria, and the lake can be listed as impaired based on phosphorus alone, in Category 5A. In this case, only one year of overwhelming exceedance is required if that year is not an extreme weather year (see Chapter 2.5 in WisCALM on *Data Requirements* for a definition of extreme weather year), and biological confirmation is not required (though can be included if available).
- *Biology Only—based on impairment of uses:* If a lake's phosphorus concentration does not exceed the criteria, but at least one biological metric is exhibiting impairment over two years, the lake can be listed for biology only. In these cases, the lake would be listed as having an impaired fish and aquatic life or recreational use under Category 5A, but the pollutant associated with this impairment may be listed as "Unknown" instead of as "Phosphorus". If it is believed that phosphorus is the causal factor in the biological impairment, the lake may be a good candidate for a more stringent site-specific phosphorus criteria.
- *TP & biology in combination—based on TP and chlorophyll exceedance:* If TP exceeds the criteria but not by 1.5 times, biological confirmation will be used to determine what listing category is appropriate.
    - If at least one of the biological response metrics is poor for at least one year, the lake should be listed as impaired for fish and aquatic life and/or recreational uses under Category 5A, with phosphorus listed as the pollutant.
    - If either insufficient biological data are available to conduct an assessment or biological data are available and do not indicate an impairment, the lake will be placed in Category 5P[19]. This category is a special category on the impaired waters list for waters exceeding TP criteria but without biological information indicating an impairment. More monitoring is needed, and/or other metrics may need to be considered. Cateogory 5P lakes may be good candidates for site-specific phosphorus criteria.

Assessment scenarios incorporating TP and biological data are listed in Table 4.

[18] For lakes an "overwhelming exceedance" is defined as 1.5 times the phosphorus criteria; for rivers/streams, an "overwhelming exceedance" is defined as 2 times the phosphorus criteria.
[19] All Category 5P waters require TMDLs, but will be given a low priority for TMDL development.

Table 4. Assessing phosphorus and biology in combination to determine impairment status and pollutant.

	Biological Response Indicators	Overall Assessment Result & EPA Listing Category	Pollutant
Meets TP criteria	None indicate impairment	Not Impaired (Fully Supporting) Category 2	NA
	One or more indicate impairment	Impaired – Biology Only (Not Supporting) Category 5A	Unknown
Exceeds TP criteria (not an overwhelming exceedance)	One or more indicate impairment	Impaired – TP & Biology (Not Supporting) Category 5A	TP
	None indicate impairment	Impaired – Exceeds TP but has insufficient or conflicting biological data (Not Supporting) Category 5P	TP
Exceeds TP criteria by an overwhelming amount	None needed	Impaired – TP Only (i.e. Overwhelming exceedance) (Not Supporting) Category 5A	TP

Dissolved Oxygen (DO)
Low DO can be used as an impairment indicator. This standard implies an activity that causes a change in DO above and beyond natural variability, or some uncontrollable factor (such as drought).

Minimum Data Requirements

a) *Seasonal Range and Sampling Frequency.* A minimum of 10 discrete values over a period of 5 years, collected on separate calendar days during the ice-free period are required from each assessment station. If more samples than the minimum are available, they will also be used in calculations unless excluded due to professional judgment.

b) *Sampling Depth.* Samples should be taken from the epilimnion. In the case of two-story lakes, samples should be taken from both the epilimnion and hypolimnion.

c) *Units.* DO values should be expressed in mg/L.

d) *Data Quality.* If data quality for any values is questionable, they should not be used for the calculations. Data should only be used from DO meters where calibration records are available, or from titration methods. (However, this information is all field-entered, so the data points are not automatically flags to indicate suspect data.)

Calculations and Exceedance Frequencies

a) *Calculations.* Data from the most recent 5-year period may be lumped together for this calculation (however, the data should all be from a single station). If 10% of values exceed DO criteria, the lake is not meeting criteria. Because low DO most commonly occurs in shallower portions of a lake, individual station data should be assessed separately to determine whether DO problems exist.

b) *Exceedance Frequency.* Compare data to the impairment threshold for DO listed in Table 5A. For all lakes except Two-Story Lakes, the threshold is less than 5 mg/L. For Two-Story Lake, the threshold is less than 5 for the epilimnion and less than 6 for the hypolimnion, where coldwater species may be found. If 10% or more of all DO values (from all assessment sites combined, cumulatively over the most recent five year period) are below the applicable thresholds, the impairment threshold is exceeded.

Macrophytes (aquatic plant metrics)

Aquatic plants respond to human disturbance (Lacoul & Freedman 2006, Wilcox 1995). Certain plant species are lost when nearshore areas are developed or when non-point source pollution, especially phosphorus, impacts water chemistry, triggering a response from aquatic plant communities. Plants can be used as a metric to signify ecological impairment, for example, due to eutrophication[20]. The department has employed a standardized point-intercept sampling method beginning in 2005 to make data more comparable across lakes and to gain lake-wide coverage of the entire aquatic plant community (Hauxwell et al. 2010, Mikulyuk et al. 2010). Methodological standardization has resulted in high among-lake comparability and robust estimations of species richness and frequency of occurrence.

In this assessment cycle, we are exploring how a combination of both multivariate and multi-metric methods can be used to assess aquatic macrophyte communities in lakes. Multivariate community analysis can be used to compare aquatic plant communities in assessment lakes to those in undisturbed reference sites. Lakes that have substantially different plant communities from reference lakes can be flagged for further investigation. The aquatic plant data from flagged systems can then be used to calculate a number of metrics that indicate human perturbation. Individual metrics can be combined into a comprehensive index score. One of these indices, called the Aquatic Macrophyte Community Index, or AMCI, decreases with increasing human disturbance. This multi-metric aquatic plant index was created by Nichols, Weber, and Shaw (2000) using data from transect-based plant surveys of Wisconsin lakes. Current analysis is underway to evaluate the component metrics of the AMCI and consider additional or alternative plant metrics that are most informative at identifying impaired lakes.

Because a waterbody's overall AMCI score reflects a wide range of stressors, WDNR researchers have determined that for purposes of impairment (303(d)) listing related to individual stressors such as phosphorus, it is more appropriate to use a combination of plant community information and individual plant metrics correlated to that stressor, instead of the overall AMCI score. For 2014, WDNR has developed protocols for assessing the following variables and metrics that correlate to elevated phosphorus levels and eutrophication impairments in Wisconsin lakes:

- Plant species abundance
- Plant community composition

[20] For 2014, plant metrics will be used primarily to assess eutrophication impacts to fish and aquatic life uses. In the future, they may also be used to assess habitat degradation or recreational use impairments.

- Relative % littoral area vegetated
- Relative % tolerant species
- Maximum depth of plant growth

Biological impairment will be analyzed using a reference condition approach. We selected a pool of reference lakes representing regional least-impacted conditions as defined by land-use at the watershed and local scale (100m shoreline buffer). The reference plant communities serve as benchmarks against which other plant communities may be compared. However, environmental factors not related to humans influence aquatic plant communities and also must be accounted for before making comparisons (Mikulyuk et al. 2011). Thus, we grouped reference lakes according to plant community composition. Lakes fell into three major groups that were best explained by latitude and substrate type (soft vs. sandy). The assessment procedure involves assigning category membership to new assessment lakes (based on latitude and substrate), and then comparing the test community to those communities in the appropriate reference group using multivariate methods (Reynoldson et al. 1995). If plant communities in comparison lakes are found to be significantly different, than an investigation into the possible sources of impairment proceeds first by evaluating the scores of individual impairment metrics.

The impairment indicated by different aspects of an aquatic plant community will vary. For example, maximum depth of plant growth (MDC) and relative frequency of tolerant species (TOL) both indicate a eutrophication impairment, while frequency of floating-leaf plants (FLOAT) signifies a habitat degradation impairment. The metrics that appear to be most strongly related to land-use disturbance are frequency of floating-leaf plants (buffer zone urban disturbance) and relative frequency of tolerant species (watershed agriculture disturbance).

For the 2014 cycle, the department has appointed an Aquatic Botanists Review Team. For lakes that biologists suspect that plant metrics may indicate impairment, the Aquatic Botanists Review Team will review the available data and make a determination based on their established protocols and best professional judgment as to whether Fish and Aquatic Life uses are impaired due to aquatic plants. Such a determination may also be used to corroborate total phosphorus exceedance.

Table 5. Fish & Aquatic Life Use impairment thresholds for lake natural communities.

Indicators	Min. Data Requirement[4]	Exceedance Frequency (see text for details)	Impairment Threshold - LAKES - Fish & Aquatic Life Use						
			Shallow			Deep			
			Headwater Drainage Lake	Lowland Drainage Lake	Seepage Lake	Headwater Drainage Lake	Lowland Drainage Lake	Seepage Lake	Two-story fishery lake
Biological indicators									
Chlorophyll *a*	3 monthly values from each of two years[3] from the period July 15 – Sept. 15	Lower bound 90%CI of the mean exceeds threshold	≥60 ug/L (≥71 TSI)	≥60 ug/L (≥71 TSI)	≥60 ug/L (≥71 TSI)	≥27 ug/L (≥63 TSI)	≥27 ug/L (≥63 TSI)	≥27 ug/L (≥63 TSI)	≥10 ug/L (≥53 TSI)
Aquatic plant metrics	*Baseline aquatic plant survey*	*NA (1 survey)*	*(Data will be reviewed by DNR's Aquatic Botanist Review Team for impairment assessments)*						
Conventional physico-chemical indicators									
Total phosphorus (TP)	3 monthly values from the period June 1 –Sept. 15	Lower bound 90%CI of the mean exceeds threshold	≥100 ug/L	≥100 ug/L	≥100 ug/L	≥60 ug/L	≥60 ug/L	≥60 ug/L	≥15 ug/L
Dissolved oxygen (DO)	10 discrete[1] epilimnetic values (ice free period, epilimnetic samples)	Greater than 10% of values	< 5 mg/L						
Temperature	20 discrete[1] values collected within a given calendar month	Greater than 10% of daily maximum or any weekly average temperature values[5] in a calendar month	See Table 4 of NR 102.25(4) of Wis. Admin. Code for acute and sub-lethal temperature criteria by calendar month for non-specific waters						
pH	10 discrete[1] values	Vary (see thresholds)	- Outside the range of 6.0-9.0 - Change >0.5 units outside natural seasonal maximum (mean) & minimum (mean) [2]						
Aquatic Toxicity-based indicators									
Acute aquatic toxicity	2 values within a 3-year period	Maximum daily concentration not exceeded more than once every 3 years	Criteria in NR 105.05 Wis. Adm. Code						
Chronic aquatic toxicity		Maximum 4-day concentration not exceeded more than once every 3 years	Criteria in NR 105.06 Wis. Adm. Code						

(1) Discrete values refer to samples collected on separate calendar days. DO, temperature and pH criteria are taken from s. NR 102.04, Wis. Adm. Code, Water Quality Standards for Wisconsin Surface Waters.
(2) Based on historical data or reference site.
(3) When used in combination with TP criteria exceedance to assess impairment, chlorophyll data from only one year is required.
(4) Smaller datasets may be considered in certain cases, such as a high magnitude of exceedance.
(5) Weekly average temperature values are calculated using the daily max values when comparing data against applicable sub-lethal criterion.

4.5 Lake Impairment Assessment: Recreational Uses

Recreational Use impairments for lakes are based primarily on both phosphorus and chlorophyll *a* (Chl *a*) levels, as Chl *a* is a measure of algal concentrations. For the 2014 assessments, the protocols for assessing both phosphorus and chlorophyll have been revised significantly from 2012. The assessments now utilize a more sophisticated statistical approach that more appropriately accounts for the variability of water quality samples. As with Fish & Aquatic Life listings, once individual metrics for eutrophication are assessed, phosphorus results should be reviewed in combination with biological response indicators such as chlorophyll to make a determination as to which listing category the lake should be placed into. This is described in Chapter 4.4 Lakes Fish & Aquatic Life, under the subheading "Determine listing categories"

Total Phosphorus (TP)
For recreational uses, TP data are assessed in the same way as described in Chapter 4.4 Lakes Fish & Aquatic Life, but the resulting 90% confidence intervals are compared to different, lower thresholds, as shown in Table 6.

Algal blooms (chlorophyll *a*)
Algae, including blue-green algae, are naturally occurring organisms found throughout the state and are an important part of Wisconsin's freshwater ecosystem. However, excessive nutrient loading (particularly phosphorus) can cause algae populations to grow rapidly under certain environmental conditions and form "blooms" that can impact water quality and pose health risks to people, pets, and livestock. Blue-green algae pose the greatest nuisance and risk to those recreating. Most species of blue-green algae are buoyant and when populations reach bloom densities, they float to the surface where they form scum layers or floating mats. In Wisconsin, blue-green algae blooms generally occur between mid-June and late September, although in rare instances, blooms have been observed in winter, even under the ice.

Algae blooms can cause many water quality problems including: a) reduced light penetration affecting the ability of macrophytes to thrive; b) discoloration of water; c) taste and odor concerns, and d) reduced DO concentrations due to massive decomposition of the cells when they die-off. Another important consequence of blue-green algae is their ability to produce naturally-occurring toxins. Effects of algal toxicity and related thresholds are discussed further in the Public Health Uses Chapter.

Calculating percent days with nuisance algal blooms and confidence intervals for Chl a
The assessment protocol for determining if Chl *a* is exceeding a recreational use threshold is significantly different in 2014 than it was in 2012. In 2012, the threshold was a concentration threshold, similar to that used for TP. For 2014, the protocol has been changed to better reflect actual impairments of recreational uses, and to better capture the variability of chlorophyll in lakes. The protocol now uses the <u>percent of days</u> during the sampling season that a lake experiences <u>nuisance algal blooms</u> as its benchmark for assessments. Nuisance algal blooms are defined as exceeding 20 ug/L Chl *a*. This was defined based on user perception surveys conducted in Minnesota. For deep lakes, the impairment threshold is 5% of days of nuisance algal blooms during the sampling season. For shallow lakes, the impairment threshold is 30% of days of nuisance algal blooms during the sampling season.

For Chl *a* recreational use assessments, the same protocols apply for data selection and calculating a grand mean as those described for chlorophyll in Chapter 4.4 Lakes Fish & Aquatic Life. However, the following statistical formula replaces that found under the subheader "Calculate confidence intervals for TP REC & FAL, & Chl FAL."

The statistical formula for Chl *a* recreational assessments determines the frequency that a lake exceeds a specific chlorophyll threshold, and also calculates the 90% confidence interval. This formula is difficult to run manually but can be done through use of a programming package such as "R" (http://www.r-project.org/). Use the following procedure to calculate the percent of days a lake is exceeding 20 ug/L chl a (P):

1. Using the chlorophyll sample values, calculate $= \frac{20-\bar{x}}{\sigma}$, where $\bar{x}$ is the sample mean and σ is the sample standard deviation.
2. Using the T table provided by the department[21], for each confidence level (lower 90%, Tlow; median, Tmed; and upper 90%, Thigh), and for the appropriate value of n (number of samples), find the value of T that is closest to the one calculated in step 1.
3. Report the value of P that is associated with the value of T that was selected in step 2.

In the absence of meeting minimum data requirements (for instance, nearshore data are available but not from the deep station), the professional judgment of the Regional Biologist should be used to consider listing any waterbody that experiences frequent and severe algal blooms where there is strong reason to believe that designated uses are impaired and nutrient levels may be contributing to such blooms. Information such as taste and odor complaints, documentation of toxin-producing blue-green algae genera, and algal cell counts can be used as justification for impairment determinations based on best professional judgment.

[21] The department can provide the appropriate T table file upon request as a CSV file (Ttable.csv).

Table 6. Recreational impairment thresholds for lake natural communities

Note: For all parameters, the assessment period is the most recent 10 year period. For TP and chlorophyll *a*, data from within the most recent 5-year period are prioritized for impairment assessments.

Indicators	Min. Data Requirement (see text for details)	Exceedance Frequency (see text for details)	Impairment Threshold - LAKES - Recreational Use						
			Shallow			Deep			
			Headwater Drainage Lake	Lowland Drainage Lake	Seepage Lake	Headwater Drainage Lake	Lowland Drainage Lake	Seepage Lake	Two-story fishery lake
Conventional physico-chemical indicators									
Total phosphorus (TP)	3 monthly values from the period June 1 –Sept. 15	Lower bound 90%CI of the mean exceeds threshold	≥40 ug/l	≥40 ug/l	≥40 ug/L	≥30 ug/L	≥30 ug/L	≥20 ug/L	≥15 ug/L
***Biological** indicators*									
Chlorophyll *a* [1]	3 monthly values from each of two years[2] from the period July 15 –Sept. 15	Lower bound 90%CI of the mean exceeds threshold	> 30% of days in sampling season have "nuisance algal blooms (> 20 ug/L)			> 5% of days in sampling season have "nuisance algal blooms" (> 20 ug/L)			
Aquatic plant metrics*	Baseline aquatic plant survey	N/A (one survey)	*(reserved until guidance available)*						

(1) While the TP impairment thresholds for the Recreational Use are based on codified criteria, the chlorophyll *a* thresholds for impairment and plant metrics assessments protocols are not codified.

(2) When used in combination with a TP dataset for impairment assessments, chlorophyll data from only one year is required.

Macrophytes (aquatic plants)
Although healthy aquatic plant communities are necessary for a good quality lake system, impacted lakes that receive high nutrient inputs may respond not with excessive algal blooms (and the associated high chl *a* values), but instead may exhibit very high macrophyte growth that is matted and densely topped out across the lake surface. This can impact recreational boating and swimming if it becomes a severe problem.

For 2014, the department has developed listing protocols based on macrophyte metrics for use in determining Fish & Aquatic Life use impairments, as described in Chapter 4.4 Lakes Fish & Aquatic Life. However, more research is needed to define how to appropriately conduct recreational use assessments based on macrophytes. WDNR recognizes the importance of developing such a protocol, and hopes to further investigate this issue through additional research and data review, for use in future listing cycles. Such research may investigate correlations between density of macrophytes or frequency of species occurrence with impacts such as inhibited recreational uses or increased issuance of Aquatic Plant Management permits.

Invasive species such as Eurasian Water Milfoil and Curly Leaf Pondweed often contribute to high macrophyte levels. However, Wisconsin does not list waters as impaired due to invasive species, as no guidance is yet available from EPA on how to do so.

Inland and Great Lakes Beaches
Many, but not all, beaches are evaluated for Recreational Uses in Wisconsin. Federal criteria for *Escherichia coli* (*E. coli*) are applicable to the open waters of the Great Lakes – including beaches. In Wisconsin, inland beaches follow the same monitoring and assessment protocol as the Great Lakes beaches. *E. coli* is a species of bacteria that serves as an indicator of the presence of fecal matter in the water – suggesting that there may be harmful bacteria, viruses, or protozoans present that elevate risk to humans.

Monitoring for *E. coli* at many public beaches along the shorelines of Lake Michigan and Lake Superior is conducted in accordance with the Beach Environmental Assessment and Coastal Health Act of 2000 (the BEACH Act). Since 2003, approximately 122 monitoring sites[22] at public beaches in Wisconsin are sampled for *E. coli* for implementation of the BEACH Act. Beaches included in the monitoring program get sampled between 1 and 4 times per week depending on the priority given to the beach. For more information on Wisconsin's Beach Program please visit: www.wibeaches.us.

Although *E. coli* may not be representative of the pathogen strains that result in illness to humans, its presence suggests that fecal matter may be in the water and that other pathogens may be present. It is often these and other pathogens that result in water borne illnesses in humans. Data from this effort are used to make decisions on which beaches are impaired – namely due to chronic closure problems due to the presence of high counts of *E. coli* bacteria.

EPA has established two different water quality criteria for *E. coli* – a single sample maximum of 235 colony forming units (cfu) /100 mL and a long-term geometric mean[23] maximum of 126 cfu/100 mL. Beach closure decisions are routinely made considering the single sample value. However, when evaluating *E. coli* data to determine if a beach should be included on the Impaired Waters List, WDNR relies on long-term data sets.

[22] A few beaches in Wisconsin have beaches large enough that multiple sites are sampled at the beach. In these cases, samples from multiple sites on one beach are often combined to make up a composite sample.
[23] A geometric mean is a measure of central tendency calculated by multiplying a series of numbers and taking the n^{th} root of the product, where n is the number of items in the series

To assess the attainment of recreational uses at Wisconsin beaches, WDNR aggregates by month all data collected from beaches during the "beach season" (defined as May 1 through September 30) over the past five years[24]. The data is aggregated by month because it more closely approximates the "five samples per month" requirement of the geometric mean criterion and recognizes that typical sampling frequencies are often less than five times per month. For example, Monthly aggregate data sets with fewer than five data points are considered insufficient for assessing recreational use support. If one or more of the monthly-aggregated geometric means exceeds the criterion of 126 cfu/100ml, the beach will be identified as not supporting its recreation use and placed on the Impaired Waters List. When a beach is included on the proposed Impaired Waters List, the pollutant is listed as *E. coli* and the impairment is identified as "Recreational Restrictions – Pathogens." WDNR will propose to remove a beach from the Impaired Waters List when the monthly-aggregated geometric means of data collected during the previous five years meet the criterion of 126 cfu/100 ml. WDNR believes this is an appropriate way of recognizing chronic risk to human health associated with recreational activities in water with long-term elevated levels of *E. coli.*

4.6 Lake Impairment Assessment: Public Health and Welfare Uses[25]

Harmful Algal Blooms- Blue Green Algal Toxin Health Risks

Algal toxins can be harmful to humans and animals alike through skin contact, inhalation, or ingestion. Some of the species commonly found in Wisconsin that produce algal toxins include *Anabaena* sp., *Aphanizomenon* sp., *Microcystis* sp., and *Planktothrix* sp. Where monitoring of blue-green algae occurs, notices are provided to local public health agencies when concentrations are presumed to exceed 100,000 cells/L. That value represents the threshold for high risk to humans as established by the World Health Organization (WHO) (Table 7). Illnesses related to blue-green algae can occur in both humans and pets. People may be exposed to these toxins through contact with the skin (e.g., when swimming), through inhalation (e.g., when motor boating or water skiing), or by swallowing contaminated water. In 2009, the Wisconsin Department of Health Services documented over 41 cases statewide of human health exposure related to blue-green algae blooms including respiratory ailments (coughing), watery eyes and rashes. Animals can be even more susceptible to risks by drinking water directly from water bodies with dense algal blooms or by licking their fur after swimming.

Biologists should use best professional judgment in determining whether the "High Risk" thresholds in Table 7 are exceeded on a regular basis. When a waterbody is proposed to be included on the Impaired Waters List due to frequent and elevated blue green algal cell counts or toxins, and data are available suggesting high TP concentrations, the Impairment should be identified as "Public Health-Harmful Algal Blooms." In the absence of meeting minimum data requirements for TP (for instance, nearshore data is available but not deep hole data), the professional judgment of the Regional Biologist should be used to consider listing any waterbody that experiences frequent and severe blue-green algal blooms or elevated levels of toxins where there is strong reason to believe that nutrient levels may be contributing to such blooms.

If data are frequently falling into the "Moderate Risk" category, the lake should be considered for Recreational Use listing based on the guidelines in that chapter.

[24] For example, the five year assessment period for the 2012 Impaired Waters List is January 1, 2006 through December 31, 2010.
[25] Although in the future, WDNR hopes to categorize impairments due specifically to Blue Green Algal Toxins under a Public Health & Welfare Use impairment category, for 2014 they will be categorized under Recreational Use Impairments.

Table 7. World Health Organization thresholds of risk associated with potential exposure to cyanotoxins.

Indicator (units)	Low Risk	Moderate Risk	High Risk
chl a (µg/L)	<10	10 - <50	>50
Cyanobacteria cell counts (cells/L)	< 20,000	20,000 - <100,000	≥ 100,000
Microcystin	<10	10 - ≤20	>20

5.0 Stream & River Classification and Assessment Methods

5.1 Stream and River Classifications

The condition of streams and rivers in Wisconsin are currently assessed for the following use designations: Fish and Aquatic Life, Recreational Use, Public Health and Welfare (Fish Consumption) and General Uses. The following provides details on the classifications and water quality goals against which waters are assessed.

Fish and Aquatic Life: Stream and River Classifications

Wisconsin's Fish and Aquatic Life (FAL) use designations for streams and rivers are categorized into the following subcategories as defined in s. NR 102.04(3), Wis. Adm. Code:

- **Coldwater (Cold) Community**: Streams capable of supporting a cold water sport fishery, or serving as a spawning area for salmonids and other cold water fish species. Representative aquatic life communities associated with these waters generally require cold temperatures and concentrations of DO that remain above 6 mg/L. Since these waters are capable of supporting natural reproduction, a minimum DO concentration of 7 mg/L is required during times of active spawning and support of early life stages of newly-hatched fish.

- **Warmwater Sport Fish (WWSF) Community**: Streams capable of supporting a warm water-dependent sport fishery. Representative aquatic life communities associated with these waters generally require cool or warm temperatures and concentrations of DO that do not drop below 5 mg/L.

- **Warmwater Forage Fish (WWFF) Community**: Streams capable of supporting a warm water-dependent forage fishery. Representative aquatic life communities associated with these waters generally require cool or warm temperatures and concentrations of DO that do not drop below 5 mg/L.

- **Limited Forage Fish (LFF) Community**: Streams capable of supporting small populations of forage fish or tolerant macroinvertebrates that are tolerant of organic pollution. Typically limited due to naturally poor water quality or habitat deficiencies. Representative aquatic life communities associated with these waters generally require warm temperatures and concentrations of DO that remain above 3 mg/L.

- **Limited Aquatic Life (LAL) Community**: Streams capable of supporting macroinvertebrates and/or occasionally fish that can tolerate organic pollution. Typically this category includes small streams with very low-flow and very limited habitat. Certain marshy ditches, concrete line-drainage channels, and other intermittent streams. Representative aquatic life communities associated with these waters are tolerant of many extreme conditions, and require concentrations of DO that remain above 1 mg/L.

Fish and aquatic life use designations for individual waters are defined in chs. NR 102 or 104, Wis. Adm. Code. In some cases, coldwater fish communities referenced in the 1980 Trout Book (Wisconsin Trout Streams – Publication 6-3600(80)) may be *codified by reference*. Waters that are not referenced in code are considered *default* FAL waters and are assumed to support either a coldwater community or warmwater community depending on water temperature and habitat.

Assignment of designated uses for the protection of fish and aquatic life has been an iterative process dating back to the late 1960's. Many of the designated uses that are included in the Wis. Adm. Code date back to the 1980's. While efforts are underway to revise FAL use subcategories, the current codified FAL use designation subcategories in ch. NR 102, Wis. Adm. Code will be used for evaluating WQS attainment status.

Natural Communities

Currently, streams and rivers are being evaluated for placement in a revised aquatic life use classification system, in which the new fish and aquatic life use subclasses are referred to as *Natural Communities*. Natural Communities are defined for streams and rivers using model-predicted flow and temperature ranges associated with specific fish and/or macroinvertebrate communities. This model, developed by the USGS and WDNR Science Services Research Staff, generated proposed stream natural communities based on a variety of base data layers at various scales, and was initially applied to the 1:100,000 scale NHD (National Hydrography Dataset) hydrography layer. The data was then extrapolated or "conflated" to the 1:24,000 scale WDNR hydrography layer (version 5). The Natural Communities data layer for Wisconsin rivers and streams identifies which fish index of biological integrity (F-IBI) to apply when assessing our waters. The following Natural Communities have been defined:

> *Macroinvertebrate* – very small, almost always intermittent streams (i.e., cease flow for part of the year, although water may remain in the channel) with a wide range of summer temperatures. No or few fish (< 25 per 100 m of wetted length) are present, but a variety of aquatic invertebrates may be common, at least seasonally.
>
> *Coldwater* – small to large perennial streams with cold summer water temperatures. Coldwater fish range from common to dominant (25-100% of individuals), transitional fish from absent to abundant (up to 75% of individuals), and warmwater fish from absent to rare (0-5% of individuals). Small-stream, medium-stream, and large-river fish range from absent to dominant (0-100% of individuals).
>
> *Cool-Cold Headwater* – small, usually perennial streams with cool to cold summer water temperatures. Coldwater fish range from absent to abundant, transitional fish from common to dominant, and warmwater fish from absent to common. Small-stream fish range from very common to dominant (50-100% of individuals), medium-stream fish from absent to very common (0-50% of individuals), and large-river fish from absent to uncommon (0-10% of individuals).
>
> *Cool-Cold Mainstem* – moderate to large but still wadeable perennial streams with cool to cold summer water temperatures. Coldwater fish range from absent to abundant, transitional fish from common to dominant, and warmwater fish from absent to common. Small-stream fish range from absent to very common, medium-stream fish from very common to dominant, and large-river fish from absent to very common.
>
> *Cool-Warm Headwater* – small, sometimes intermittent streams with cool to warm summer temperatures. Coldwater fish range from absent to common, transitional fish from common to dominant, and warmwater fish from absent to abundant. Small-stream fish range from very common to dominant, medium-stream fish from absent to very common, and large-river fish from absent to uncommon.

Cool-Warm Mainstem – moderate to large but still wadeable perennial streams with cool to warm summer temperatures. Coldwater fish range from absent to common, transitional fish from common to dominant, and warmwater fish from absent to abundant. Small-stream fish range from absent to very common, medium-stream fish from very common to dominant, and large-river fish from absent to very common.

Warm headwater – small, usually intermittent streams with warm summer temperatures. Coldwater fish range from absent to rare, transitional fish from absent to common, and warmwater fish from abundant to dominant. Small-stream fish range from very common to dominant, medium-stream fish from absent to very common, and large-river fish from absent to uncommon.

Warm mainstem – moderate to large but still wadeable perennial streams with warm summer temperatures. Coldwater fish range from absent to rare, transitional fish from absent to common, and warmwater fish from abundant to dominant. Small-stream fish range from absent to very common, medium-stream fish from very common to dominant, and large-river fish from absent to very common.

Large rivers – non-wadeable large to very-large rivers. Summer water temperatures are almost always cool-warm or warm, although reaches are identified based strictly on flow. Coldwater fish range from absent to rare, transitional fish from absent to common, and warmwater fish from abundant to dominant. Small-stream fish range from absent to uncommon, medium-stream fish from absent to common, and large-river fish from abundant to dominant.

5.2 Stream and River General Condition Assessment

Fish and Aquatic Life General Assessments

WDNR uses biological indices, including fish indices of biological integrity (F-IBI) and the macroinvertebrate index of biological integrity (M-IBI), to determine whether current water quality conditions support the Fish and Aquatic Life designated use.

Fish Indices of Biological Integrity

Multiple, peer-reviewed F-IBIs have been developed by WDNR research staff and are used to assess the biological health and quality of fish assemblages of streams and rivers (Lyons, Wang, and Simonson 1996; Lyons 1992, 2003, 2006, and 2012). F-IBIs have been customized to account for differences in stream morphology, water temperature and fish species associated with rivers and streams. A fish IBI has not been developed for any of the small streams lacking sufficient perennial flow to support a fish community. The indices use a large statewide database of standardized fish assemblage surveys from numerous reaches with different levels of human impact. An objective procedure was used to select and score the metrics that compose the various F-IBIs, choosing metrics that represent a variety of the structural, compositional, and functional attributes of fish assemblages (Table 8).

Table 8. Fish Indices of Biological Integrity for Wisconsin streams and rivers.

	Cold F-IBI (Lyons et. al, 1996)	**Warm F-IBI** (Lyons, 1992)	**Small F-IBI** (Lyons, 2006)	**Large River F-IBI** (Lyons et. al, 2001)	**Cool-Warm F-IBI** (Lyons, 2012)	**Cool-Cold F-IBI** (Lyons, 2012)
Temperature	Maximum daily mean <22° C	Maximum daily mean >22° C	Maximum daily mean >22° C	N/A	Maximum daily mean 22.6–24.6 °C	Maximum daily mean 20.7–22.5 °C
Applicable Stream Size & Location	Streams of any size or watershed area	Wadeable streams of a width between 2.5m and 50m, and depth of at least ~1.25m	Streams with watershed areas that are $4km^2$ to $41km^2$	Rivers with at least 3km of contiguous, non-wadeable channel	Scoring criteria depend on the watershed area (“large” is > 200 km^2 and “small” is ≤ 200 km^2) and latitude (“north” > 44.6◦N and “south” is ≤ 44.6◦N)	Scoring criteria depend on the watershed area (“large” is > 200 km2 and “small” is ≤ 200 km2) and latitude (“north” > 44.6◦N and “south” is ≤ 44.6◦N)
Individual Metrics	a) # intolerant species b) % tolerant species c) % top carnivore species d) % native or exotic stenothermal coldwater or coolwater species, e) % salmonid individuals that are brook trout	a) # native species b) # darter species c) # sucker species d) # sunfish species e) # intolerant species f) % tolerant species g) Percent omnivores h) % insectivores i) % top carnivores j) % simple Hthophils k) # of individuals per $300m^2$ l) % diseased fish	a) # native species b) # intolerant species c) # minnow species d) # headwater species e) Total catch per 100m, excluding tolerant species f) Catch per 100 m of brook stickleback g) % diseased fish	a) Weight Biomass PUE b) # native species c) # sucker species d) # intolerant species e) # riverine species f) % diseased fish g) % riverine h) % lithophils i) % insectivore j) % round suckers	a) # native minnow species b) # intolerant species c) % tolerants d) # benthic invertivore species e) % omnivores	a) # darter, madtom and sculpin species b) # coolwater species c) # intolerant species d) % tolerant species e) % generalist feeders

Macroinvertebrate Indices of Biological Integrity
Data derived from aquatic macroinvertebrate samples, combined with stream habitat and fish assemblages, provide valuable information on the physical, chemical and biological condition of streams. Most aquatic macroinvertebrates live for one or more years in streams, reflecting various environmental stressors over time. Since the majority of aquatic invertebrates are limited in mobility, they are good indicators of localized conditions, upstream land use impacts and water quality degradation.

WDNR uses the M-IBI developed by Weigel (2003) to assess wadeable streams. The M-IBI is composed of various metrics used to interpret macroinvertebrate sample data. The M-IBI was developed and validated for cold and warm water wadeable streams and cannot be used as an assessment tool for non-wadeable rivers or ephemeral streams. The following metrics are included in the M-IBI:

- Species richness
- Ephemeroptera–Plecoptera– Trichoptera (EPT)
- Mean Pollution Tolerance Value
- Proportion of Depositional Taxa
- Proportion of Diptera (Dipt)
- Proportion of Chironomidae (Chir)
- Proportion of Shredders (Shr)
- Proportion of Scrapers (Scr)
- Proportion of Gatherers (Gath)
- Proportion of Isopoda (Isop)
- Proportion of Amphipoda

A new macroinvertebrate IBI has been developed, validated, and applied to assess nonwadeable rivers (Weigel and Dimick 2011). Hester–Dendy artificial substrates were used to conduct a standardized macroinvertebrate survey at 100 sites on 38 nonwadeable rivers in Wisconsin. Ten metrics that represent macroinvertebrate assemblage structure, composition, and function constitute the IBI:

- Number of Insecta taxa
- Number of EPT taxa
- Proportion of Insecta individuals
- Proportion of intolerant EPT individuals
- Proportion of tolerant Chironomidae individuals
- Proportion of gatherer individuals
- Proportion of scraper individuals
- Proportion of individuals from the dominant 3 taxa
- Mean Pollution Tolerance Value
- Number of unique functional trait niches

Fish and macroinvertebrate data are used to calculate the appropriate F-IBI and M-IBI scores. Biological data collected within the last ten years are assessed. General biological condition assessments require at least one F-IBI value or one M-IBI value, whereas at least two values of a particular index are required for impairment assessments. Due to strong temporal variations in biological assemblage characteristics at degraded sites, more samples and a longer time frame are needed to determine biotic integrity at sites with human impacts than is needed at least-impacted sites (Lyons, et. al 2001). Natural Community classifications are used to determine which biological index to apply (Table 9).

Table 9. Modeled water temperature and flow criteria used to predict Natural Communities in healthy Wisconsin streams and the primary index of biotic integrity (IBI) for bioassessment associated with each Natural Community.

Natural Community	Maximum Daily Mean Water Temperature (˚F)	Annual 90% Exceedence Flow (ft^3/s)	Index of Biotic Integrity
Macroinvertebrate	Any	0.0 – 0.03	Macroinvertebrate
Coldwater	< 69.3	0.03 – 150	Coldwater Fish
Cool-Cold Headwater	69.3 - 72.5	0.03 – 3.0	Small-Stream (Intermittent) Fish
Cool-Cold Mainstem	69.3 - 72.5	3.0 – 150	Cool-Cold Transition (Coolwater) Fish
Cool-Warm Headwater	72.6 - 76.3	0.03 – 3.0	Small-Stream (Intermittent) Fish
Cool-Warm Mainstem	72.6 - 76.3	3.0 – 150	Cool-Warm Transition (Coolwater) Fish
Warm Headwater	> 76.3	0.03 – 3.0	Small-Stream (Intermittent) Fish
Warm Mainstem	> 76.3	3.0 – 150	Warmwater Fish
Large River	Any	> 150	River Fish

The biological indices respond to watershed scale impacts of agricultural and urban land uses, local riparian stressors, nutrient enrichment, and instream habitat degradation including sedimentation and scouring. In general, as the rate of stream degradation increases, a corresponding decrease in the number of environmentally-sensitive species and an increase in environmentally tolerant species are observed. These changes in aquatic community composition are scored relative to a reference or "least-impacted" condition, and are placed in a condition category based on the resulting score. The condition categories (excellent, good, fair, poor) and corresponding F-IBI scores are shown in Table 10, and the wadeable M-IBI and nonwadeable river M-IBI thresholds are given in Tables 11 and 12, respectively. To determine the biological condition of streams and rivers for assessments, the F-IBI or M-IBI values should be compared against thresholds established for each natural community class.

For general condition assessments, all waters scoring in the excellent, good, or fair categories are considered supporting the FAL use, unless corroborating physical or chemical data exceed impairment thresholds. Waters scoring in the poor condition category based on general assessments using one bioassessment result (available from Tier 1 monitoring) are flagged for follow-up (Tier 2) monitoring.

Table 10. Condition category thresholds for applicable fish indices of biotic integrity (IBI).

Natural Community	Fish IBI Type	Fish IBI	Condition Category
Coldwater	Coldwater Fish	81-100	Excellent
		51-80	Good
		21-50	Fair
		0-20	Poor
Cool-Cold or Cool-Warm Headwater	Small-Stream (Intermittent) Fish	91-100	Excellent
		61-90	Good
		31-60	Fair
		0-30	Poor
Cool-Cold Mainstem	Cool-Cold Transition Fish	61-100	Excellent
		41-60	Good
		21-40	Fair
		0-20	Poor
Cool-Warm Mainstem	Cool-Warm Transition Fish	61-100	Excellent
		41-60	Good
		21-40	Fair
		0-20	Poor
Warm Headwater	Small-Stream (Intermittent) Fish	91-100	Excellent
		61-90	Good
		31-60	Fair
		0-30	Poor
Warm Mainstem	Warmwater Fish	66-100	Excellent
		51-65	Good
		31-50	Fair
		0-30	Poor
Large River	River Fish	81-100	Excellent
		61-80	Good
		41-60	Fair
		0-40	Poor

Table 11. Condition category thresholds for wadeable stream macroinvertebrate index of biotic integrity.

Wadeable Stream M-IBI Thresholds	*Condition Category*
> 7.5	Excellent
5.0-7.4	Good
2.5-4.9	Fair
< 2.5	Poor

Table 12. Condition category thresholds for nonwadeable river macroinvertebrate index of biotic integrity.

River M-IBI Thresholds	*Condition Category*
>75	Excellent
50-75	Good
25-49	Fair
<25	Poor

5.3 Stream and River Impairment Assessment: Fish & Aquatic Life Uses

To make an impairment assessment, all available data over the last 10-year period are reviewed.
If a stream or river general assessment category is 'poor', an impairment assessment is conducted. Data up to the past decade, preferably from within the past five years, can be used when conditions are confirmed to be stable throughout the assessment time period. Biological data alone can be used to list a water as impaired, as long as minimum data requirements are met. At least two samples of one biological assemblage (fish or macroinvertebrates) collected in different calendar years are required to assess biological condition for impairment listings. However, if corroborating water quality or physical habitat data exists, one fish survey or one macroinvertebrate sample may be sufficient for impairment listing decisions. For example, if the biological condition category is 'poor' based on the IBI value, and minimum total phosphorous sampling requirements are met and the TP concentrations exceed the impairment threshold, the water would be listed for "degraded biological community" impairment with the pollutant total phosphorus listed as the "cause" of the impairment.

Additional targeted monitoring may be needed to identify a particular pollutant/impairment combination and could include supplemental physical and chemical data, as well as biological data, at additional monitoring sites to obtain adequate coverage of extent of impairment (Table 13). WDNR Biologists have knowledge of the factors that influence community response in rivers and streams. Those insights should be considered when selecting indicators to collect or when scheduling supplemental monitoring. Potential stressors and habitat surveys can help choose the appropriate parameters to be monitored and evaluated to confirm the impairment and to define the associated pollutant. Field collection, preservation and storage should follow procedures outlined in the WDNR Field Procedures Manual and laboratory analysis should follow standard methods (Wisconsin State Lab of Hygiene, 1993).

Table 13. Additional parameters for river & stream impairment assessments.

Indicator	Indicator
Alkalinity	Nitrogen – (Nitrate & Nitrite)
Ammonia*	Organic Compounds*
Biochemical Oxygen Demand	Periphyton
Chlorides*	pH*
Dissolved Oxygen*	Phosphorus – Ortho
Exotic Species – Abundance	Phosphorus – Total*
Exotic Species – Presence/Absence	Sediment Chemistry
Flow	Solids – Total Suspended
Habitat – Qualitative	Solids – Settleable
Habitat – Quantitative	Specific Conductivity
Hardness	Temperature%
Heavy Metals*	Toxicity – Ambient*
Land Use	Toxicity – Sediment
Nitrogen – Total Kjeldahl	Transparency

* = Numeric Water Quality Criteria are available in chs. NR 102 or 105, Wis. Adm. Code

Specific Protocols and Indicator Thresholds for Impairment Decisions

Biological Indicators
As in general condition assessments, biological indicators are also used to assess attainment of WQS and determine whether the fish and aquatic life uses are supported. Section NR 102.01(2) of Wis. Adm. Code explains the goal of WQS is to "protect the use of water resource for all lawful purposes… which includes the protection of public health and welfare and the present and prospective uses of all waters of the state for public and private water supplies, propagation of fish and other aquatic life and wild and domestic

animals, domestic and recreational purposes, and agricultural, commercial, industrial, and other legitimate uses. Chapter 102.04(1)d Wis. Adm. Code provides narrative standards for the protection of fish and other aquatic life in surface waters, stating "Substances in concentrations or combinations which are toxic or harmful to humans shall not be present in amounts found to be of public health significance, nor shall substances be present in amounts which are acutely harmful to animal, plant or aquatic life." For streams and rivers, attainment of the narrative biological standards is assessed using the fish and macro-invertebrate indices described in the previous section. Biological indicator data collected from two or more sampling visits for a particular assessment unit (i.e. stream segment) are considered sufficient data to assess attainment of the narrative biological standards (Table 14A). The general condition category threshold for "poor" condition is used as the benchmark for evaluating attainment of WQS.

Total Phosphorus

For streams and rivers, TP can be linked as a pollutant causing biological impairment using WDNR's sampling protocol, which has been developed consistent with considerations of seasonality, timing and frequency of sample collection used by USGS for development of the TP criteria (s. NR 102.06(3), Wis. Adm. Code). Waters should be sampled monthly over a 6-month period from May through October, ideally within the same year. Each sample should be collected approximately 30 days apart, with no samples collected within 15 days of one another. If more than one sample is available per month, the sample closest to mid-month should be used in the analysis. If one or more monthly samples are missed within a year, additional samples may be collected in subsequent years corresponding with the missed months (e.g., if July and August samples were not collected in the first year, they could be collected in the second year to make a complete data set). If multiple years of data are available, the three most recent years of data should be used. TP data collected for study-specific purposes as part of a targeted monitoring design (e.g., storm event sampling or targeted flow regimes) are not appropriate for assessment of attainment of the applicable TP water quality criterion.

A parametric statistical approach is employed to assess stream TP data against the applicable water quality criterion found in NR 102.06 of Wis. Admin. Code. This approach involves the calculation of a 90% confidence limit around the median of a TP sample dataset. A confidence limit is calculated using measures of sample size and variation to suggest with a specified level of certainty that the true population statistic (e.g. median) falls within a specified range of values. This statistical approach is described in Gibbons (2003).

The 90% lower and upper confidence limits (LCL and UCL, respectively) on the 50th percentile (median) of a normal distribution are computed as:

$$LCL = \bar{x} - Ks$$
$$UCL = \bar{x} + Ks$$

> Where $\bar{x}$ is the sample mean, s is the observed sample standard deviation, and K is the one-sided normal tolerance limit factor for the median with 90% confidence and *n* samples.

WDNR uses automated database assessment packages to perform the calculations for sampling stations that meet the minimum data requirements for assessment purposes. Along with the automated assessment packages, an Excel spreadsheet template is also available for performing the calculations manually. Manual calculations of the statistical values may be required to assess data that is not in the SWIMS database.

If the LCL of the phosphorus dataset from a particular stream site exceeds the applicable criterion, and those data were representative of normal weather and hydrology, then the corresponding stream segment is considered to be exceeding the TP criteria. Two assessment paths lead to listing a stream or river for the pollutant TP in the standard impaired waters category, Category 5A. If the LCL exceeds the applicable TP criterion by two-fold (i.e., "overwhelming exceedance"), then biological confirmation of impairment is not required. However, if the LCL exceeds the criterion less than two-fold (under normal weather and hydrologic conditions), a F-IBI or M-IBI score indicating 'poor' biological condition sufficiently corroborates the FAL use impairment. Waters that exceed TP criteria, but biological data are not available or the biological assessment does not indicate impairment, will be placed in an impaired waters subcategory, Category 5P. These waters are assigned a high priority for biological data collection to determine appropriate future management actions. All Category 5P waters require TMDLs, but will be given a low priority for TMDL development. These TP-related impairment listing scenarios are summarized in Table 4 of Section 4.4.

Other physical/chemical indicators

For other physical/chemical parameters listed in Table 14, monitoring data are evaluated against minimum data requirements, specific thresholds and allowable exceedance frequencies as indicated in the table. If readily available data for the parameters listed are evaluated and determined to be insufficient (i.e. does not meet minimum data quantity requirements), but the limited data indicates a potential use impairment, the waterbody may be a designated as a "Watch Water," and assigned a higher priority for monitoring in the near future.

Table 14. Fish and aquatic life use impairment thresholds for rivers/streams.

Parameters	Minimum Data Requirement[26]	Exceedance Frequency	Cold Waters	Warm Waters	Limited Forage Fish	Limited Aquatic Life
Conventional physical and chemical indicators						
Dissolved Oxygen	3 days of continuous measurements (no less than 1 sample per hour) in July or August collected from each of 2 separate calendar years.	Greater than 10% of values	<6.0 mg/L and <7.0 mg/L during spawning season	<5.0 mg/L	<3.0 mg/L	<1.0 mg/L
Temperature	10 discrete daily values[27] or days of continuous temperature data[28] collected within a given calendar month to assess against acute and sub-lethal criteria, respectively.	Greater than 10% of daily maximum values or any weekly average temperature value[29] in a calendar month exceeds acute criteria or sub-lethal criteria, respectively.	See Table 2 of NR 102.25(2) of Wis. Admin. Code for acute and sub-lethal temperature criteria by calendar month for non-specific waters			
pH	10 discrete daily values[27]	Greater than 10% of values within a continuous sampling period or for instantaneous w/in season	Outside the range of 6.0 to 9.0 standard units (SU), or change is > 0.5 SU outside natural seasonal maximum (mean) and minimum (mean)			
Total Phosphorus[30]	6 samples monthly from May through October	Lower 90% confidence interval of the sample median exceeds threshold	≥0.100 mg/l for rivers; ≥0.075 mg/l for streams			
Biological indicators						
Fish IBI	1 value when used in combination with TP data. For a standalone bio-assessment, 1 value from each of 2 years within 5 years	1 value when used in combination with TP data. For a standalone FAL listing, average value from 2 samples across 2 years	See "poor" condition thresholds in Table 10			
Macroinvertebrate IBI	1 value when used in combination with TP data. For standalone bio-assessment, 1 value from each of 2 years within 5 years	1 value when used in combination with TP data. For standalone FAL listing, average value from 2 samples across 2 years	See "poor" condition thresholds in Tables 11 and 12			

Note: Data are evaluated from within the most recent 10 year period for all parameters.

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[26] Smaller datasets may be considered in certain cases, such as a high magnitude of exceedance.

[27] Discrete values refer to samples collected on separate calendar days.

[28] To assess against the applicable sub-lethal temperature criterion, continuous temperature data should be collected at a frequency of no less than one sample per hour with a continuous recording thermistor.

[29] Weekly average temperature values are calculated using the daily max values when comparing data against applicable sub-lethal criterion.

[30] One 'poor' F-IBI or one 'poor' M-IBI is also required to corroborate the impairment of the FAL use for standard impaired waters Category 5A listings. Streams exceeding TP criteria alone will be placed in an impaired waters subcategory, Category 5P.

Table 14 (continued). Fish and aquatic life use impairment thresholds for rivers/streams.

Aquatic Toxicity-Based indicators			
Acute aquatic toxicity indicators	**Minimum Data Requirement**	**Exceedance Frequency**	**Criteria Table Reference**
Cadmium*, Chromium$^{(3+)}$*, Copper*, Lead*, Nickel*, Zinc*, Pentachlorophenol, and Ammonia *(*total recoverable form)*	2 values within a 3-year period	Maximum daily concentration not exceeded more than once every 3 years	Criteria in NR 105.05 Wis. Adm. Code
Arsenic$^{(+3)}$*, Chromium$^{(+6)}$*, Mercury$^{(+2)}$*, free Cyanide, Chloride, Chlorine (total residual), Gamma - BHC, Dieldrin, Endrin, Toxaphene, Chlorpyrifos, and Parathion (**total recoverable form)*			Criteria in NR 105.05 Wis. Adm. Code
Chronic aquatic toxicity indicators			
Cadmium*, Chromium$^{(3+)}$ *, Copper*, Lead*, Nickel*, Zinc*, Ammonia and Pentachlorophenol *(*total recoverable form)*	2 values within a 3-year period	Maximum 4-day average concentration not exceeded more than once every 3 years	Criteria in NR 105.06 Wis. Adm. Code
Arsenic$^{(+3)}$*, Chromium$^{(+6)}$*, Mercury$^{(+2)}$*, free Cyanide, Chloride, Chlorine (total residual), Dieldrin, Endrin, and Parathion (**total recoverable form)*			Criteria in NR 105.06 Wis. Adm. Code

5.4 Stream and River Impairment Assessment: Recreational Uses

Federal criteria for *E. coli* were developed after consideration of risk to the swimming public. All of the data used to establish the federal criteria were collected from swimming beaches. In general, flowing rivers and streams in Wisconsin do not provide comparable recreational activities for full body immersion. For those water bodies, WDNR utilizes that the long-standing water quality criterion for fecal coliform that is reflected in s. NR 102.04(5), Wis. Adm. Code. That section reads:

> (a) *Bacteriological guidelines.* The membrane filter fecal coliform count may not exceed 200 per 100 ml as a geometric mean based on not less than 5 samples per month, nor exceed 400 per 100 ml in more than 10% of all samples during any month.

When a flowing stretch of a river or stream is included on the proposed Impaired Waters List, the pollutant is listed as fecal coliform and the impairment is identified as "Recreational Restrictions – Pathogens." In many instances where fecal coliform counts are high, *E. coli* data or other pathogen data are also collected for streams and rivers and may be used in lieu of or supplementary to fecal coliform data to make best professional judgment decisions to list or not list the waterbody as impaired.

6.0 Public Health and Welfare Uses[31] applicable to all waterbody types

6.1 Fish Consumption Use Assessment

Waterbodies may be designated as impaired on the 303(d) list based on the level of fish consumption advice, which, in Wisconsin, is due primarily to mercury, PCBs, dioxin and furan congeners, and Perfluoroctane sulfonate (PFOS). In 1998, 241 waters were added to the 303(d) list in Category 5B[32], "Waters Impaired by Atmospheric Deposition of Mercury," because mercury-based fish consumption advisories had been issued for these specific waterbodies based on advisory protocols then used by Wisconsin (1985 and 1986 Mercury Protocols).

In 2001, Wisconsin adopted a statewide general advisory that applies to all (non-Great Lakes) waters of the state based on statewide distribution of mercury in fish and species differences in mercury concentrations. The statewide general advisory eliminated the need for many of the pre-2001 advisories because the equivalent of more stringent advice now applied through the general advisory. In addition to the statewide general advisory, some waters still required more stringent advice or exceptions to the general advisory. Exceptions to the general advice apply to some species of fish from specific waters where higher concentrations of mercury, PCBs or other chemicals require advice more stringent than the general advisory.

Since 2002, the 303(d) list has been updated based on changes made to the list of specific advisory waters. However, most of the pre-2001 specific advisory waters remain on the 303(d) list until re-sampling of these waterbodies occurs to confirm that the general advisory is adequate. If new data collected from a pre-2001 advisory water indicates that an exception to the general advisory is not necessary, the waterbody would be removed from the 303(d) list.

[31] For the 2014 listing cycle, impairments related to algal toxins will be listed as Recreational Use impairments (see Lakes Assessment chapter). In the future, WDNR may associate impairment causes such as fish tissue contamination, contaminated sediment, and algal toxins with impairments to the Public Health and Welfare Use.
[32] See Chapter 8 for an explanation of Integrated Report Assessment Categories.

For the 2012 impaired waters update, a waterbody will be proposed for removal from the 303(d) list when the most recent advisory update indicates that only the statewide general advisory is necessary for concentrations of bioaccumulating chemicals that are of concern in Wisconsin fish. The waters defined as impaired waters are those with specific contaminant data for game and panfish species that require advice more stringent than the statewide general advice based on examination of data in conjunction with WDNR of Health Services. Appendix B lists the fish tissue contaminant thresholds that are used when developing fish consumption advisories.

Specific waters will be proposed for de-listing where fish samples are collected and tested for the appropriate chemicals and where the general statewide advisory is determined to be adequate and exceptions are not necessary based on an evaluation of the concentrations of mercury, PCBs, dioxin/furans, or other chemicals using Wisconsin's fish advisory protocols. The general fish consumption advisory will still apply to these waters, but they will no longer be included on the 303(d) list.

Wisconsin Departments of Natural Resources and Health Services jointly manage the fish contaminant monitoring and advisory programs. The monitoring strategy for fish contaminants varies by the pollutant and the waterbody (see Wisconsin's Water Division Monitoring Strategy). WDNR fisheries staff conducts the fish sampling supported by a variety of fisheries funds. The Wisconsin State Laboratory of Hygiene supports most chemical analyses through general revenue and an agreement with the WDNR. Some EPA funds are used for supplies, lab and freezer rentals, advisory publications, and special analyses.

More information about the specific consumption advisory can be found in the publication: Choose Wisely, A Healthy Guide for Eating Fish in Wisconsin (PUB-FH-824 2010 or subsequent years.) It is available on line at http://WDNR.wi.gov/fish/consumption/.

6.2 Contaminated Sediments

Waterbodies that have sediment deposits that are known to have toxic substances that exceed state water quality criteria for ambient water (as specified in ch. NR 105, Wis. Adm. Code) will be included on the Impaired Waters List. These waters may be identified through various monitoring activities, including routine water quality monitoring, sediment analysis, and collection of fish tissue. In addition to a comparison to the water quality criteria found in ch. NR 105, Wis. Adm. Code, WDNR compares the concentrations of commonly found, in place contaminants to the values outlined in a sediment quality guidance document *Consensus-Based Sediment Quality Guidelines, WT PUB- 732, 2003* (See Appendix C). *http://www.WDNR.state.wi.us/org/water/wm/sms/documents.html*. The guidance was developed through an assimilation of results from multiple published effects-based toxicity testing to freshwater benthos, and serves as part of a tiered approach to evaluating potential ecological and human health risks at sites under evaluation for various reasons.

7.0 Making a Decision to List or Delist Waterbodies

Once data have been assessed to determine whether any parameters indicate impairment of a waterbody, a decision to list a waterbody as impaired or to delist a waterbody should be made. There are several nuances to this decision that are discussed in this chapter. These include resolution of conflicting results from different parameters on a waterbody, identification of which Use Designations are impaired, determination of the appropriate EPA category, and identification of "Causes" and "Sources" of impairment.

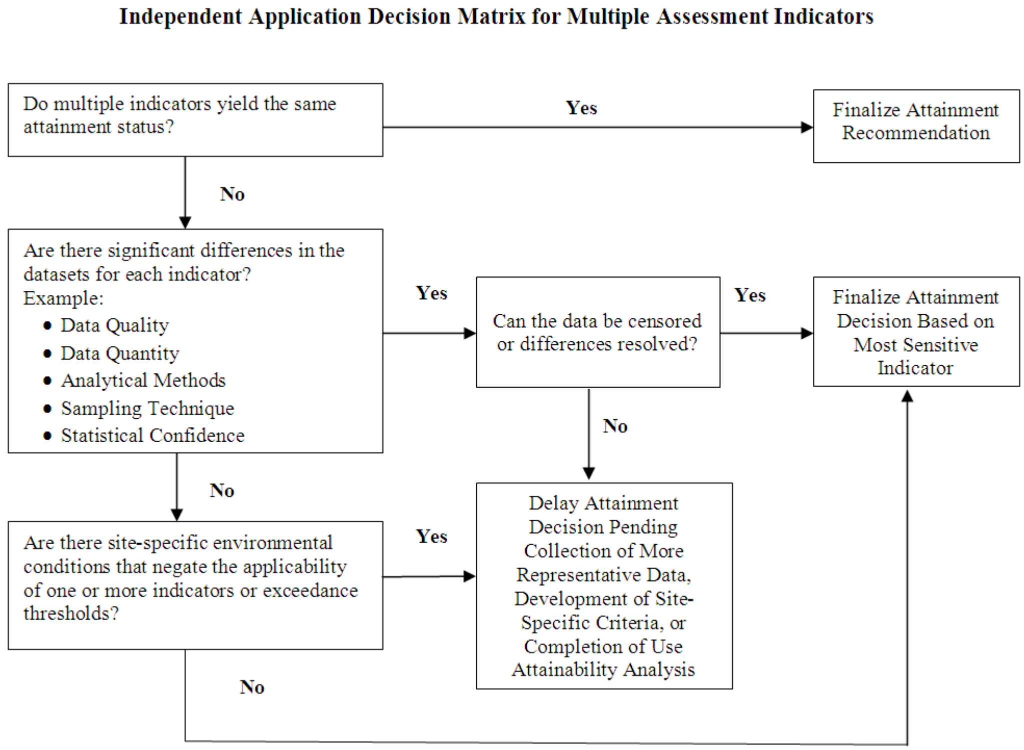
When minimum data requirements are met, an attainment decision should be made and documented. When a decision is made to not list a waterbody due to insufficient data, where limited data show criteria excursions, the water is identified as a "Watch Water" and prioritized highly for future monitoring allow in order to collect sufficient data for future assessment. All assessment results and impaired waters listing details are documented in the WATERS database.

7.1 Independent Applicability & Tools to Resolve Data Conflicts

Under Federal guidance, a water shall be listed on the Impaired Waters List if data is reflective of current conditions, data has met minimum data requirements, and the water does not meet WQS, including water quality criteria, designated uses, and/or antidegradation. This decision philosophy is referred to as *independent applicability,* consistent with the Clean Water Act that protects biological, chemical, and physical integrity of surface waters. However, EPA recognizes that there are certain situations in which factors beyond a strict interpretation of Independent Applicability should be considered to make the most appropriate listing decision. When assessing whether a water is attaining narrative WQS, for example, a suite of indicators are often used. Accordingly, EPA allows states to formulate specific decision rules pertaining to circumstances under which one type of parameter should be given a greater 'weight' than others. Wisconsin has developed decision rules that use a hierarchy of indicators for certain parameters, which are described within the Lakes and Rivers & Streams chapters of this guidance document.

If one of the WQS are not met, but multiple data sets produce conflicting results (some indicating impairment and some not), WDNR staff should review all available data to assist in making an attainment decision. There are several factors biologists may use to resolve these differences to arrive at a listing decision. A decision matrix is described in Figure 14 to describe the process for *not* making attainment decisions using independent application. Cases where this process is used will be rare and should be well documented for that water in the WATERS database.

Figure 14. Independent Application Matrix



Data quality differences

If one parameter indicates impairment but another does not, differences between the two data sets in data quality, data quantity, analytical methods, sampling technique or statistical confidence may provide reason to weight one set of data more heavily than another.

Site-specific factors

Natural background levels of a pollutant may be higher than impairment thresholds or uncontrollable factors may cause an exceedance of WQS. In these circumstances, WDNR will determine whether criteria exceedance are reasonably expected to be due to natural or uncontrollable causes, as defined in the “Six Factors” of Use Attainability Analysis (40 CFR 131.10(g)). If assessment documentation supports that impairment is due to natural or uncontrollable factors, a Use Attainability Analysis (UAA) should be pursued to modify the Designated Use and/or associated criteria. However, a water with suspected naturally occurring pollutant levels that exceed applicable water quality criteria should be placed on the Impaired Waters List under Category 5C, until the appropriate designated use and/or site-specific water quality criteria have been approved by WDNR and EPA. Category 5C waters are those that are identified as impaired, but the cause of the impairment may be attributed to natural or uncontrollable source(s) (see Table 15).

Weight of Evidence

In certain cases where data sets conflict with one another, states may apply a “weight of evidence” approach. This approach helps define the extent of the problem based on how it impacts the Designated Use, and allows biologists to consider aspects of the data that might indicate whether one data set should be weighted more greatly than another.

In all cases, Department staff will look for corroborating information, such as the various habitat and biological indices and water chemistry data. If the suite of available data does not suggest an evident impairment, then the water will not be listed, but will be recommended for additional monitoring as resources allow. WDNR will provide a rationale for those cases where data are available that show that a water quality criterion has been exceeded, but the water has not been recommended for the impaired waters list. In those cases, the indicator has not reached the magnitude, duration or frequency to warrant placing a waterbody on the list or the available data from a particular indicator are not representative of current conditions.

Hierarchy of Indicators
In some situations, *a hierarchy of the indicators may be appropriate*. For example, biological indicators (e.g., fish or macroinvertebrate IBI) for assessment of the fish & aquatic life use may have precedence over physical or chemical indicators in the impairment decision process, because they are direct measures of health of aquatic life. However, this hierarchical approach should be used with caution, knowing that exceedance of chemical indicators may correspond to a more recent event that was not reflected in the biological community data due to differences in collection periods or delays in community response. In such a case, a decision to rely on a hierarchical approach would be inappropriate.

When assessing waters against the applicable phosphorus criteria, biological data are used in combination with phosphorus data to determine whether the fish and aquatic life use is currently impaired. If biological impairment is observed, the water is placed in the standard impaired waters category (5A). If the water exceeds phosphorus criteria but biological impairment is not observed, the water is placed in an impaired waters subcategory (5P) that is given a lower priority for management actions, until biological impairment is confirmed.

7.2 Professional Judgment

WDNR staff most familiar with a waterbody should be directly involved in the assessment decision. Staff knowledge and experience along with the factors that influence water quality should be considered when reviewing and interpreting available data. Professional staff should explore a myriad of issues to determine the most relevant and appropriate data to use for attainment decisions, including: data quality, frequency and magnitude of exceedances, weather and flow conditions during sample collection, anthropogenic or natural influences on water quality in the watershed, etc. If any available data is not used because of professional judgment, clear documentation of the reasons for doing so should be included in the final attainment decision. Again, whether a waterbody is listed as impaired, or the decision has been made not to list a waterbody, all decisions should be *well documented* within the database and future management recommendations will be noted on waters that were not listed (for example, a formal use designation change is needed in order to list the water as impaired, and a recommendation would be made in WATERS to reflect this need).

Two specific review stages occur during the assessment process when regional water resource biologists review the preliminary assessment results. The first review is a data review of the automated database assessment packages. The package results include a series of downloadable reports and spreadsheet outputs for some assessment parameters, which are provided to biologists for review. At that time, reviewers may document justification for a different assessment result based on data quality, additional data and/or waterbody classification errors. After incorporating all assessment and listing modifications from the data review, a Professional Judgment Team will review the draft assessment results and make recommendations for any needed modifications. The following questions may be considered during the professional judgment review stage:

- Are the data from appropriate weather and flow conditions, or are they limited to critical hydrological regimes (low and high flows)? If data are available only from extreme weather years (as defined in Section 2.5), should that dataset be supplemented with data from current conditions before making an assessment decision?

- Are data representative of current water quality conditions?
- Have land uses or point sources changed substantially since the data were collected?
- If the minimum data requirements are not met, do the limited data provide overwhelming evidence of impairment (e.g., phosphorus dataset does not meet minimum data requirements, but biological impairment has been documented or the phosphorus criterion is exceeded by double).

7.3 Threatened Waters

Wisconsin recognizes *threatened* waters as defined by the United States Environmental Protection Agency (EPA):

> *Any waterbody of the United States that currently attains water quality standards, but for which existing and readily available data and information on adverse declining trends indicate that water quality standards will likely be exceeded by the time the next list of impaired or threatened waterbodies is required to be submitted to EPA.*

Waters identified as *threatened* waters become a formal part of the Impaired Waters List, with all of the ramifications associated with impaired waters. Currently no guidance exists on how to formally list *threatened* waters as impaired, waters that fall into this category may be evaluated on a case-by-case basis. A biologist would have to provide sufficient data and information (e.g., 5-10 years of data and multiple samples per year to run a regression analysis) that clearly shows a "declining trend" to predict that the water would be impaired by the next listing cycle. If such significant data exists, the water could be considered for listing as threatened on the Impaired Waters List.

7.4 Watch Waters

Watch Waters are those for which limited data indicate potential impairment, but insufficient data are available to make a final impairment decision, and, therefore, are identified for further monitoring. These waters are not included on the Impaired Waters List due of circumstances warranting further observation or evaluation.

For example, a water may be designated as a Watch Water if water quality data indicating impairment are were collected from unrepresentative "extreme weather" periods, as defined in Section 2.5, resulting in insufficient data to assess. Watch Water status is also designated when phosphorus data are assessed for a particular water but a "clear" decision cannot be made (i.e. 90^{th} percent confidence interval of the phosphorus sample concentration data overlaps the criterion). WisCALM guidance defines a "clear" exceedance of the phosphorus criteria as the lower 90^{th} percent confidence interval of a phosphorus sample concentration dataset that exceeds the applicable criterion. Conversely, the phosphorus criteria are "clearly met" when the upper 90^{th} percent confidence interval of the phosphorus sample concentration data is below the applicable criterion.

7.5 Identifying Sources of Impairment

When a water is deemed impaired, the potential source(s) causing the impairment should be identified. Impairment sources affect which parameters are monitored, what model should be used for analysis and what type of restoration activities would be best on that individual water. In the WATERS database, under the "WDNR Impaired Waters Category" sources may be entered. Some possible sources of impairment include:

Atmospheric Deposition: Waters with fish consumption advisories (FCAs) caused by atmospheric deposition of mercury. To a very limited extent, it may include waters with advisories due to polychlorinated biphenyls (PCBs) where no discrete contaminated sediment deposits exist.

Contaminated Sediment: Waters identified through various monitoring activities, sediment core analysis, and collection of fish tissue that exceed ambient water quality criteria for toxics as specified in ch. NR 105, Wis. Adm. Code. In addition this may include waters where contaminated sediments contain pollutant concentrations that will cause "probable effects" in biological organisms based on guidelines outlined in the "Consensus-Based Sediment Quality Guidelines: Recommendations for Use and Application (2002).

Physical Habitat: Waters where codified uses are not being met due to a physical structure, such as a dam (e.g., a downstream segment is deemed impaired due to the presence of a dam preventing fish movement).

Point Source Dominated: Waters are categorized as point source dominated when the impairment is a result of a current discharge from an existing point source. The Wisconsin Pollutant Discharge Elimination System (WPDES) Permit Program issues and evaluates permits for point sources to assure the attainment of standards at the time of permit issuance. Existing laws and administrative rules including the WQS and WPDES permit rules preclude the issuance of a permit if it will not attain WQS. Waters in this category are likely between permit cycles, or may have obtained a variance to the WQS under current law.

Nonpoint Source (NPS) Dominated: Waters in which the impairment is a result of nonpoint source runoff, including urban stormwater runoff.

Nonpoint Source/Point Source Blend: Waters are placed in this category when impairments exists due to both point source contributions and nonpoint source runoff. Listing a waterbody which is impacted by a point source does not imply that the source is not meeting all the requirements in its discharge permit, but only indicates that a TMDL is needed to determine relative contributions by each of the sources and what additional requirements may be needed.

7.6 De-listing Impaired Waters

Waters and/or associated pollutants and impairments are de-listed from the state's impaired waters list when the state determines and the EPA approves that the waters are no longer impaired or a particular pollutant impairment combination should be removed. A water will not be delisted until all previously listed pollutant/impairment combinations have been removed because applicable WQS are attained. WDNR proposes to de-list a waterbody and/or associated pollutants and impairments from the Impaired Waters List when contemporary, representative, and high quality data warrant de-listing. However, when a change to a water quality standard (e.g. site-specific criteria) has been approved by EPA and the waterbody now meets the revised criterion, WDNR may propose to remove the water and/or associated pollutants and impairments from future lists.

Water No Longer Impaired

WDNR de-lists waters that have been restored. New monitoring data will be collected through Tier 3 monitoring to evaluate the response of the waterbody to some sort of implementation or restoration strategy. Waters will be assessed through the same process identified as listing a waterbody on the 303(d) Impaired Waters List and must meet WQS to be removed from the list.

If a portion of a previously listed water is later determined to be no longer impaired, while other portions remain impaired, the originally listed water may be subdivided into multiple assessment units to account for these differences in attainment status.

Water Listing Validation Found No Impairment

WDNR has identified some waters on historical Impaired Waters Lists that may be inappropriately listed. Common reasons include improper documentation of a past assessment, misidentification of a waterbody, and/or incorrect description of the reach and its specific location within a watershed. In those cases, contemporary information will be documented and WDNR may propose to de-list those waters if the most recent assessment indicates all designated uses are achieved.

EPA Approved TMDL
When EPA approves a TMDL, the water pollutants covered by the TMDL are proposed for removal from EPA-approved list of impaired waters that require a TMDL (Category 5 waters). However, the water is still considered impaired until applicable WQS have been met. Waterbodies having completed TMDLs are moved to Category 4A (Table 15). Once the water is restored and meets applicable water quality criteria, it may be moved to Category 2.

7.7 Decision Documentation

A primary goal of the WDNR is to document all impaired waters decisions, verify the current impaired waters list, and make this information accessible to the public. It is critical that WDNR staff fully document their impaired waters listing recommendations, supporting materials, and justification of their decisions, including any professional judgment used to support those decisions. As a part of this process, it is also important to document assessment decisions for waterbodies that were evaluated but deemed fully supporting assessed uses. The WATERS data system for monitoring and assessment data provides WDNR staff with a systematic location and process for documenting assessment decisions.

Data contained in these data systems are available for the public via the WDNR Surface Water Data Viewer. Information such as monitoring stations, Impaired Waters, WPDES permits, etc. can be accessed from this site. WDNR also maintains dynamic webpages created for Impaired Waters where the public can find water quality monitoring data, pollutants/impairments of concern, TMDL status, and possible management solutions for improving the waterbody. The Impaired Waters Search Tool may be accessed at following website: http://dnr.wi.gov/water/impairedSearch.aspx.

Assessments of non-conventional parameters or those that deviate from standard WisCALM guidance should be documented on the standardized documentation form (Appendix A) and include a justification or case-specific reason for diverging from the assessment guidance. An electronic documentation form is available on request; please send requests to DNRImpairedWaters@wisconsin.gov.

8.0 Integrated Report Listing Categories

One of the elements of the Integrated Report (IR) is defining IR listing categories (Table 15) for each waterbody or assessment unit to communicate work conducted under the use designation, assessment and restoration elements of the WQS program. Wisconsin's IR listing categories loosely follow federal categories identified in the 2008 EPA Integrated Reporting Guidance document.

Table 15. Integrated Report (IR) Listing Categories

IR Category	How Categories Are Used in Wisconsin
Category 1	All designated uses are met, no use is threatened, and the anti-degradation policy is supported. This category requires that all designated uses have been assessed for a given water.
Category 2	Available information indicates one or more designated uses are met. This category is applied to waters that have been assessed and considered fully meeting one or more designated uses and is usually applied in Wisconsin to waters that have been restored and removed from the impaired waters list.
Category 3	There is insufficient available data and/or information to assess whether a specific designated use is being met or if the anti-degradation policy is supported. This category is also used for situations where the state has not yet had time or resources to analyze available data.
Category 4: Waters where a Total Maximum Daily Load (TMDL) is approved by EPA or not required.	
Category 4A	All TMDLs needed for attainment of water quality standards have been approved or established by EPA. This does not mean that all other designated uses have been evaluated and found to be meeting their designated use.
Category 4B	Required control measures are expected to achieve attainment of water quality standards in a reasonable period of time. Environmental Accountability Projects may be proposed as an alternative to TMDL development.
Category 4C	A waterbody where the impairment is not caused by a pollutant. Pollution is defined by EPA as the human-made or human-induced alteration of the chemical, physical, biological, and radiological integrity of water (Section 502(19)).
Category 5: Waters where a TMDL is required.	
Category 5A	Available information indicates that at least one designated use is not met or is threatened and/or the anti-degradation policy is not supported, and one or more TMDLs are still needed.
Category 5B	Available information indicates that atmospheric deposition of mercury has caused the impairment of the water. The water is listed for a specific advisory and no in-water source is known other than atmospheric deposition.
Category 5C	Available information indicates that non-attainment of water quality standards may be caused by naturally occurring or irreversible human-induced conditions.
Category 5P	Available information indicates that the applicable total phosphorus criteria are exceeded; however, biological impairment has not been demonstrated (either because bioassessment shows no impairment or because bioassessment data are not available).
Category 5W	Available information indicates that water quality standards are not met; however, the development of a TMDL for the pollutant of concern is a low priority because the impaired water is included in a watershed area addressed by at least one of the following WDNR-approved watershed plans: adaptive management plan, adaptive management pilot project, lake management plan, or Clean Water Act Section 319-funded watershed plan (i.e. nine key elements plan).

Placing Assessment Units in Categories
Evaluated waters are placed in Category 3 unless sufficient data or information is available to move the water from a Category 3 to a different group. Waters that meet one or more designated uses -- and have no uses impaired will be included in Category 2. For example, if a waterbody was previously listed as impaired, but, subsequently restored and removed from the impaired waters list, it may then be placed in Category 2. This category cannot be used for situations in which one or more use designations have been restored but other use designations remain impaired. Waters will be placed in Category 2 after WisCALM guidance has been applied and the water has been fully assessed through an impaired waters de-listing process and determined to be meeting applicable WQS.

WDNR assigns a listing category to both the overall water and individual pollutant/water combinations in our WATERS database. If one pollutant listing has been removed from a water (e.g., because the applicable criteria are now met for that pollutant) but additional pollutant listing(s) remain, the overall waterbody will remain in an impaired water category (i.e., Category 5) until all pollutant listings have been removed. Categories are also assigned to pollutant/water combinations, in part, to allow WDNR to track the TMDL status of each pollutant listing. For example, for a waterbody with multiple pollutant listings, Category 4a is assigned to pollutant listings when a TMDL has been developed, while other pollutant listings that do not have a completed TMDL are assigned to Category 5.

Moving Assessment Units between Categories
Waters are moved from one category to another during updates to the assessment database by water quality biologists and program coordinators. Once an assessment has been conducted the water will be moved from Category 3, to the updated category. This process usually occurs once a year during the update of the state's water assessments during basin plan updates.

Assessment Units with multiple pollutant/impairment listings
Wisconsin uses one category per water, as well as a category for each pollutant/impairment listing combination. Because of this, the waterbody is placed in the more protective or restrictive category available. For example, if a waterbody is listed for two use impairments (e.g. recreation and fish and aquatic life) and one of the two remain impaired while the other is restored, the waterbody will remain in an impaired water category (i.e., Category 5).

8.1 Priority Ranking for TMDL Development

Waters on the Impaired Waters List will be ranked by priority for Total Maximum Daily Load (TMDL) development. A TMDL is an analysis that determines how much of a pollutant a waterbody can assimilate before it exceeds WQS. Federal law requires that TMDLs be developed for impaired waters.

Waters are ranked "high," "medium" or "low." Rankings are evaluated during each listing cycle to determine if TMDL development can be completed based on staff and fiscal resources. If a TMDL is in development, we will rank the waterbody as a "high" priority. A ranking of "medium" indicates that information is currently being gathered that may be used for future TMDL development. All Category 5B waters (waters impaired by atmospheric deposition of mercury) will be assigned a "medium" priority. A ranking of "low" indicates that a TMDL will be completed in the future.

The following factors are considered when selecting waters for TMDL development:

- **Availability of information**: Large amounts of data are needed to develop a TMDL. Some waters already have some water quality data that can be used while others have little to no data to determine pollutant sources or loading. Waters with readily available data will more likely be a candidate for TMDL development within two to five years and assigned a "medium" or "high" priority ranking.

- **Likelihood to respond:** WDNR may consider the likelihood of the water to respond to management actions when assigning a rank.
- **Severity of the impairment:** WDNR will also consider the severity of the impairment in assigning a priority. In some cases, extreme conditions may be present that need attention more quickly than those that are not so extreme. Waters with frequent fish kills or acute toxicity issues are examples of this concern.
- **Public health concerns:** Waters with issues that may affect human health can be considered "high" priority if development and implementation of a TMDL can result in improving water quality.

Environmental Accountability Projects (EAPs)

Alternatives to a TMDL can be prepared for waters on the 303(d) list. These alternatives are referred to as "Environmental Accountability Projects" or EAPs. These are any planned implementation actions on the impaired water that will result in that water meeting WQS. EAPs are commonly used when the source of an impairment and the appropriate management action are readily identifiable. EAP listings are designated when of the sources and pathways of pollutants do not require a TMDL analysis to identify management actions. Wisconsin currently has several projects that may have an EAP analysis prepared to address specific pollutants and impairments.

9.0 Public Participation

WDNR recognizes the importance of public involvement in the assessment, restoration and protection of the state's water resources. Public involvement in the development of the state's Impaired Waters List is also required by the Clean Water Act. Several opportunities are provided for public comment on the water quality assessments related to the development of the Impaired Waters List and Integrated Report as it is developed, including the following:

- Calls for data as public noticed by WDNR.
- Statewide public informational meetings to discuss the draft list of impaired waters and the WisCALM document used to determine impairments.
- Informal meetings, as resources allow, with interested parties.
- Draft 305(b) report and 303(d) list as public noticed by WDNR with request for comments.
- Supporting assessment documentation provided upon request.
- Public comments must be sent to WDNR during the formal comment period to be considered in the listing decision submittal. However, comments may be sent to WDNR or directly to EPA about WDNR's Integrated Report at anytime during the process.

9.1 Requests for Data from the Public

The WDNR provides an opportunity for the public, partners and stakeholders to submit water quality datasets for inclusion in assessment of waters against water quality standards for the Integrated Report of Water Quality. For the 2014 listing cycle, public data was solicited during January and February, 2013.

9.2 Submittal of Wisconsin's Integrated Report to U.S. EPA

Wisconsin will provide the EPA with an integrated dataset, a narrative report, associated spatial data files, and a list of updates to the state's 2014 Impaired Waters List on or before April 1, 2014. When this occurs, the WDNR will post the final submittal package on the agency's website for public informational purposes.

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APPENDIX A. 2014 Impaired Waters Assessment Documentation Form

2014 Impaired Waters Documentation Sheet

Author:		Date Prepared:
Waterbody Name:		Segment:
WADRS ID:	WBIC:	Use i-SWDV (CRTL + Click) to find ID numbers

Choose from the following to indicate what you are recommending:

_____ Proposed new **impaired water** listing

_____ Proposed new **watch water** listing

_____ Proposed changes for water already on 303(d) list (check type of change below) ➔ **TMDL ID #:** _________

_____ Proposed change to existing list (new pollutants, impairments, mileages, etc.)

_____ Proposed for de-listing

_____ General 303(d) documentation for water already on list

Description of waterbody segment

Start Mile: End Mile: Total miles: Lake Acres:	Detail (describe segment using road crossings, convergence with other waterbodies, etc.):

Use Designation Categories	List use designation & data source for each category.
Current (Existing) Fish & Aquatic Life Use:	
Attainable (Potential) Fish & Aquatic Life Use:	
Designated (Codified) Fish & Aquatic Life Use:	

Is it supporting its FAL Attainable Use? _____ Fully Supporting _____ Not Supporting _____ Not Assessed
Is it supporting its Recreational Use? _____ Fully Supporting _____ Not Supporting _____ Not Assessed

Does a *Specific* Fish Consumption Advisory Exist? _____ Yes _____ No _____ Don't know

If so, what is the specific advisory:

Pollutants & Impairments

Pollutants: (Place an X next to all pollutants that you are recommending for listing or de-listing, or "watch water" monitoring needs.)

Phosphorus	Sediment	Bacteria	PAHs	PCBs
NH_3 (Ammonia)	Thermal	Hg	Creosote	Metals
Unknown	Other Pollutants:			

Impairments: (Place an X next to all impairments that you are recommending for listing, de-listing, or "watch water" monitoring needs.)		
Degraded Habitat	Eutrophication	Temperature
Contaminated Fish Tissue	Chronic Toxicity	Aquatic Toxicity
Unknown	Degraded Biological Community	

Specific causes of impairment: (Describe to the best of your ability what you think is contributing to the impairment.)

Information is based on:

Monitoring data collected on/after January 1, 2003? _____ YES _____ NO

If 'NO' then provide justification for using data from the long term record:

Monitoring & Listing Data

Monitoring Study, Date, Results. List water quality exceedances indicating magnitude, duration and frequency (attach additional sheets, if needed).

Monitoring Studies:

Exceedances:

Stations:

Parameters:

Database where data is stored (Fish Database, SWIMS, FishSED, Personal PC):

Narrative on why you are proposing this waterbody to be listed or de-listed?

List and attach any additional reports, updated watershed tables, analyses etc. including use designation survey.

1.

2.

3.

4.

APPENDIX B. Summary of Fish Tissue Criteria for Fish Consumption Advice

Summary of Fish Tissue Criteria for Fish Consumption Advice in Wisconsin 2008.

Summary of Mercury Advisory Guidelines (Rfd = 0.3 ug/kg/day and 0.1 ug/kg/day)

PPM **//---Statewide Safe Eating Guidelines-----------------------------------//** //--------Site Specific Only--------------//

	Unrestricted*	1 meal/week	1 meal/month	do not eat	1 meal/week	1 meal/month
men and older women	**<0.16**	**0.16-0.65**	**>0.65**		site specific ave >0.22 and max >0.33	site specific ave >0.65 and max >0.95
	panfish, bullheads, and inland trout	gamefish and other species	muskies		panfish, bullheads, and inland trout	gamefish and other species at a site ave >0.65
	Unrestricted	**1 meal/week**	**1 meal/month**	**do not eat**	**1 meal/month**	**do not eat**
Children and women of childbearing age	**<0.05**	**0.05 - 0.22**	**0.22-0.95**	**>0.95**	site specific ave >0.22 and max >0.33	site specific ave >0.65 and max >0.95
		panfish, bullheads, and inland trout	gamefish and other species	muskies	panfish, bullheads, and inland trout	gamefish and other species

Informational Item - Update on change in the fish consumption adivsory for mercury. February 2001. Department of Natural Resources. Natural Resources Board Agenda Item (Green Sheet). Also, 2007 Mercury Addendum.

Summary of PCB Advisory Guidelines (HPV = 0.05 ug/kg/day) **General vs Site Specific**

PPM

GL Tissue Criteria	Unrestricted	1 meal/week	1 meal/month		6 meals/yr	do not eat
Panfish, inland trout, bullheads	≤0.05 for GLs (General advice for inland waters)	0.06-0.22	0.22-1.0		>1- 1.99 ppm	≥ 2 ppm
Gamefish and others		0.06-0.22 for GLs (General advice for inland waters)				

Protocol For a uniform Great Lake Sport Fish Consumption Advisory. Great Lakes Sport Fish Advisory Task Force. September 1993.

Summary of Dioxin TEC Advisory Guidelines

	do not eat
sum only furan and dioxin congeners x EPA HH TEFs for total TEC	> 10 (ng/kg) ppt dioxin equvalents

June 20, 1990. Henry Anderson, MD, Department of Health and Human Services. Memo to Jay Hochmuth. Department of Natural Resources.

Summary of Chlordane Advisory Guidelines (HPV = 0.15 ug/kg/d -)

	Unrestricted	1 meal/week	1 meal/month		6 meals/yr	do not eat
Panfish, inland trout, bullheads	≤0.16 for GLs (General advice for inland waters)	0.16-0.66	0.66-2.82		2.83-5.62	>5.62 ppm
Gamefish and others		0.16-0.66 for GLs (General advice for inland waters)				

Hornshaw. 1999 Discussion Paper for Chlordane HPV. ILEPA.

Summary of PFOS advisory Guidelines

GL Tissue Criteria	Unrestricted	1 meal/week	1 meal/month		6 meals/yr	do not eat
Panfish, inland trout, bullheads	≤40 ppb for GLs (General advice for inland waters)	40-200 ppb	200-800 ppb			>800 ppb
Gamefish and others		40 - 200 pb for GLs (General advice for inland waters)				

APPENDIX C. Consensus-Based Sediment Quality Guidelines Recommendations for Use & Application

Consensus-Based Sediment Quality Guidelines

Recommendations for Use & Application

Interim Guidance

Developed by the
Contaminated Sediment Standing Team

December 2003

WT-732 2003

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Box 7921
Madison, WI 53707

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Guidance Status

This guidance will be updated as needed. Comments and concerns may be sent to "Guidance Revisions" – RR/3, WDNR, P.O. Box 7921, Madison, WI, 53707

Notice

This document is intended solely as guidance and does not contain any mandatory requirements except where requirements found in statute or administrative rule are referenced. This guidance does not establish legal rights or obligations and is not finally determinative of any of the issues addressed. This guidance does not create any rights enforceable by any party in litigation with the State of Wisconsin or the Department of Natural Resources. Any regulatory decisions made by the Department of Natural Resources will be made by applying the governing statutes and administrative rules to the relevant facts.

Consensus - Based Sediment Quality Guidelines; Recommendations for Use & Application

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Consensus - Based Sediment Quality Guidelines; Recommendations for Use & Application

1. Overview

- Wisconsin DNR needs effects-based (i.e., empirical) sediment quality guidelines (SQGs) for commonly found, in place contaminants to serve as benchmark values for making comparisons to the concentrations of contaminant levels in sediments at sites under evaluation for various reasons (e.g., NR 347 dredging projects, degree and extent studies, screening level ecological risk assessments). There is a need for these values on lower assessment tiers and on a screening level basis and for other objectives during different phases of a site assessment.

- In the last few years, a number of entities have generated effects-based SQGs for some of the more widely measured contaminant metal and organic chemical compounds. Most of the guidelines have focused on effects to benthic-dwelling species. Watershed program staff have used some of the guidelines for evaluating sediment quality at initial or lower tiers in the assessment process for the sediment quality at sites.

- The most recent development in sediment quality guidelines is where the effect-level concentrations from several guidelines of similar narrative intent are combined through averaging to yield consensus-based lower and upper effect values for contaminants of concern (e.g., MacDonald *et al*. 2000a). The consensus-based values have been evaluated for their reliability in predicting toxicity in sediments by using matching sediment chemistry and toxicity data from field studies. The results of the reliability evaluation showed that most of the consensus-based values for individual contaminants provide an accurate basis for predicting the presence or absence of toxicity (MacDonald et al. 2000a). To predict the toxicity for mixtures of various contaminants in sediments, the concentration of each contaminant is divided by its corresponding probable effect concentration (PEC). The resulting values are called PEC-Quotients (PEC-Q). The individual PEC-Qs are summed and divided by the number of PEC-Qs to yield a mean PEC-Q. Using relationships derived from existing databases, the mean PEC-Q value can be used to predict the toxicity of a mixture of contaminants in a sediment sample. The appendix provides further explanation and examples of calculating and combining PEC-Q values.

- The CBSQGs as developed only involve effects to benthic macroinvertebrate species. A large amount of databases from toxicological research have established the cause and effect or correlations of sediment contaminants to benthic organism and benthic community assessment endpoints. The guidelines do not consider the potential for bioaccumulation in aquatic organisms and subsequent food chain transfers and effects to humans or wildlife that consume the upper food chain organisms. For the most part where noncarcinogenic or nonbioaccumulative organic chemicals are involved, the guidelines should be protective of human health and wildlife concerns. Where bioaccumulative compounds such as PCBs and methyl mercury are involved, protection of human health or wildlife-based endpoints could result in more restrictive sediment concentrations than contained in the CBSQGs. Where these bioaccumulative compounds are involved, the CBSQGs need to be used in conjunction with other tools, such as human health and ecological risk assessments, bioaccumulation-based guidelines, bioaccumulation studies, and tissue residue guidelines to evaluate the direct toxicity and upper food chain effects of these compounds. Food chain models will need to be used to estimate safe levels of contaminants in sediments that will not result in accumulated levels in upper food chain organisms that exceed toxicity and tissue reference values.

- There are a number of program needs and uses for sediment quality guidelines during a tiered assessment process for a site under investigation related to further investigative and management decisions. For consistency sake, we recommend that the consensus-based SQGs (CBSQGs) as currently developed by MacDonald et al. (2000a) be utilized in appropriate situations by all Department programs for screening sediment quality data to help estimate the likelihood of toxicity, as staff evaluate the available information in order to make case-by-case investigative and management decisions for a site. For chemicals for which CBSQGs are not available, we recommend utilizing the most reliable of other effects-based freshwater SQGs that have been published in the scientific literature or developed by WDNR or other regulatory entities. In the SQG tables that follow, these latter values are included and identified as to source. In most cases, the guidelines will need to be backed by additional sampling and field studies at sites under investigation to support the guideline-predicted biological effects.

- The MacDonald et al. (2000a) CBSQGs have a lower (threshold effect concentration - TEC) and upper (probable effect concentration - PEC) effect level at which toxicity to benthic-dwelling organisms are predicted to be unlikely and probable, respectively. There is an incremental increase in toxicity as the contaminant concentrations increase between the TEC and PEC concentrations, although specific numerical values relating to the degree of toxicity can't be derived. Based on the ranges of concentration related to the TEC and PEC values, we have developed a qualitative descriptor system to be used to provide a common basis of expressing relative levels of concern with increasing contaminant concentrations. The resulting levels of concern can be used to rank and prioritize sites for additional investigation phases. The midpoint effect concentration (MEC) is a concentration midway between the TEC and PEC concentrations.

Level of Concern	Threshold Effect Concentration (TEC)	Level of Concern	Midpoint Effect Concentration (MEC)	Level of Concern	Probable Effect Concentration (PEC)	Level of Concern
Level 1	**From CBSQGs**	**Level 2**	**TEC + PEC / 2 = MEC**	**Level 3**	**From CBSQGs**	**Level 4**
≤ TEC		**> TEC ≤ MEC**		**> MEC ≤ PEC**		**> PEC**

- Development of sediment quality guidelines is an evolving science. As additional SQGs with applicability to Wisconsin sites and reliability in predicting toxicity are developed, they in turn should be evaluated for possible replacement of the CBSQGs as appropriate. There is a need to continually reexamine the appropriate use of SQGs as management tools and to refine uses of SQGs to better predict toxicity and/or biological community impairment (Fairey et al. 2001). Given the 1) variable environmental and site-specific factors that control the sequestering, release, and bioavailability of contaminants in sediments, 2) the effects of varying mixtures of sediment contaminants, and 3) the variable sensitivities and exposure and uptake routes of benthic macroinvertebrates to contaminants, there is a continued need for guidelines to be supported by site-specific field studies. Along with numerical guidelines, biological criteria based on specific toxicity tests and identified endpoints (e.g., mortality, growth, and reproduction to the test organisms) and benthic community study metrics should be established and used, as appropriate, in evaluating sediment quality. Levels of acceptable reductions in the endpoints (e.g., no more than 20% reduction [$p < 0.05$] in endpoint response compared to the reference site or control site results in toxicity tests) that can be extrapolated to have ecological relevance for the survival of populations in the field should be established (Lawrence, 1999; Michelsen, 1999; Chapman *et al*. 1997; Suter, 1996; and Suter and Tsao, 1996) and used in the evaluation and management decisions for a contaminated sediment site.

2. Introduction

Over the past several years, different entities including several states, Canadian provinces, U.S. EPA, and various researchers have each developed sets of effects-based SQGs. The guidelines were generally developed using empirical approaches that established databases that related a range of effects (e.g. reduced survival, growth, or reproduction of benthic macroinvertebrate organisms) to a range of increasing concentrations of individual sediment-associated contaminants. The guidelines generally established two concentration levels based on effects - a lower effect level at which no or minimal effects are predicted and an upper effect concentration level at which adverse effects are highly probable or will frequently be seen. The focus for all the sets of guidelines was primarily on developing concentrations that would be protective of the majority of bottom dwelling species that reside on or in the sediments and sediment pore water. The developed guidelines generally do not consider the food chain aspects of such bioaccumulative compounds as methyl mercury and the nonpolar organic compounds (e.g., PCBs) in terms of effects to humans or wildlife.

During the early-1990's, the sediment staff within the Water Quality Standards Section of the Bureau of Watershed Management had initially used effects-based guidelines developed by the province of Ontario in Canada (Persaud *et al*.1993) and NOAA (1991) in doing screening level assessments of sediment quality for various sediment projects (e.g., NR 347 assessments and in relationship to site investigations conducted at a number of sites). In 1996, based on the studies of contaminated sediments in the Great Lakes, U.S. EPA (Ingersoll *et al*. 1996a, 1996b) produced a set of sediment quality guidelines that Water Program staff incorporated into doing assessments along with the above two sets of guidelines. The Ontario and U.S. EPA guidelines are relevant because they were developed based on databases from studies involving benthic macroinvertebrate species and sites from the Great Lakes region. Since the U.S. EPA guidelines were published, several other sets of guidelines have been developed and published (MacDonald and MacFarlane, 1999 and CCME, 1999).

The most recent development in SQGs is the consensus-based SQGs (CBSQGs) in which the geometric mean of several sets of SQGs of similar narrative intent have been integrated to yield "consensus based" lower (threshold effect concentration - TEC) and upper (probable effect concentration - PEC) effect levels (MacDonald *et al*. 2000a, 2000b ; Swartz, 1999). The CBSQGs of MacDonald *et al.* (2000a) have been adopted for use as sediment quality targets in the St. Louis River Area of concern (Crane *et al*. 2000). Prior to publication of the above consensus-based guidelines in the literature, Water Program staff used the consensus-based approach to develop sediment quality guidelines for a number of metals based on averaging the effect levels from several sets of guidelines. The latter sediment quality objectives are now being superceded by our recommendation that the CBSQGs of MacDonald et al. (2000a) be used for all future sediment quality assessments.

3. Recommendations On the Type of Sediment Quality Guidelines To Be Used

For the sake of consistency on a statewide basis in doing initial screenings of sediment quality in the lower tiers of a site assessment and for other uses, it is recommended that:

1) The CBSQGs as developed by MacDonald et al. (2000a) for the protection of benthic organisms should be considered for use by all evaluators;
2) Reliable effect-based freshwater sediment quality guidelines published in the scientific literature or in Water Quality Standards Section development memos should be used for contaminants for which CBSQGs are not available; and
3) Because points 1 and 2 above principally involve protective levels for benthic organisms, other approaches such as food chain modeling and back calculating from acceptable fish tissue levels should be used to establish protective levels of bioaccumulative contaminants in sediments for ecological receptors and humans. Water Quality Standards Section staff tentatively plan to develop a separate technical paper that lists the approaches available and calculation methods of each approach to derive concentrations of contaminants in sediments that would be protective of humans and ecological receptors such as birds and wildlife.

4. The Uses of Sediment Quality Guidelines

As discussed above, there is a need for effects-based sediment SQGs for commonly found contaminants in order to compare to the concentrations that may be in the sediments of a site under study. There is a need for these values on a screening level basis and for other needs during different phases of a site assessment. The uses for CBSQGs include:

1) To assess the quality of prospective dredged materials (NR 347 dredging projects) related to potential effects both in place, during removal activities, and at the completion of removal activities. The possible impacts of residual contaminant levels left exposed at the project depth and/or in the side walls at the project boundaries also need to be evaluated.

2) To screen study site contaminant concentrations to evaluate the relative degree of potential risks and impacts to sediment dwelling species.

3) To identify and to help prioritize sites for additional studies based on the relative degree and extent of contamination, size of contaminated deposits, and potential risks to benthic receptors. These steps can allow for a systematic basis for prioritorizing sites for allocation of available funding and resources for further monitoring.

4) To evaluate the need to collect additional sediment chemistry data, based on initial screening results, and determine the need to do a concurrent collection of biological data (e.g., toxicity testing and macroinvertebrate community studies) in a second study phase to more adequately characterize the degree and extent of contamination. The biological studies would attempt to validate if the CBSQGs are accurate predictors of toxicity and impacts to the benthic community related to the contaminant concentrations found at a site.

5) As toxicity benchmarks in the staged processes associated with screening level ecological risk assessments and the problem formulation stage of baseline ecological risk assessments (Crane *et al.* 2000; Ingersoll *et al*. 1997; U.S. EPA, 1997; WDNR, 1992). Use of the CBSQGs

as benchmarks for toxicity screening serves to 1) estimate the likelihood that a particular ecological risk exists, 2) helps identify the need for site-specific data collection efforts, and 3) helps to focus site-specific baseline ecological risk assessments.

6) As one line of evidence where multiple lines of evidence are used to support decision-making activities for a site in a weight-of-evidence approach. No single line of evidence would be used to drive decision-making. Each line of evidence should be evaluated for the 1) adequacy and quality of the data, 2) degree and type of uncertainty associated with the evidence, and 3) relationship of the evidence to the potential degree of impact being estimated. All of the lines of evidence will be integrated to characterize risk based on: 1) concurrence of all line of evidence results 2) preponderance, 3) magnitude, 4) extent, and 5) strength of relationships between the exposure and the effects data.

7) The process for assessing sediment quality as it relates to identifying surface water issues will be based on the tiered assessment framework established by the Department's Contaminated Sediment Standing Team (WDNR, 2001). The tiered framework utilizes numerical CBSQGs in the lower tiers and moves to more comprehensive, structured risk-based assessments in the higher tiers. The diversity of different types of sediment assessments and objectives calls for the need for a flexible framework with options for assessing sediment quality. More information is developed in successive tiers until it can be determined that enough information is available to adequately assess the sediment quality related to biological effects. Reasons for conducting risk-based studies at higher assessment tiers may include 1) the complexity of the interactions of the aquatic ecosystem and the contaminant stressors, 2) diverse mixtures of contaminants may be present at a site, 3) outstanding exposure issues where a risk assessment will allow realistic use of information about the natural history of a species such as foraging areas, breeding times, and migration patterns (Moore et al. 1998), and/or 4) there are unresolved issues with regard to potential human or ecological exposures. A formal risk assessment is not something that needs to be conducted at every sediment site under assessment. The appropriate risk-based studies may need to be designed and carried out at higher assessment tiers. As needed, site-specific studies can progress to effects-based testing and risk-based studies of various designs and scope. Guidance for carrying out such risk-based studies are contained in WDNR guidance documents (1992a; 1992b) and a number of U.S. EPA guidance documents (e.g. U.S. EPA, 1998).

8) The CBSQGs should not be used on a stand-alone basis to establish cleanup levels or for sediment management decision making. However, in certain situations, with the agreement of all parties involved in overseeing remediation and those responsible for remediating a contaminated sediment site, the CBSQG values deemed to be protective of the site receptors can be used as the remediation objective for a site (at or approaching the lower effect or threshold effect levels for the contaminant of concern). An example of the latter application was at Gruber's Grove Bay on the Wisconsin River, which was contaminated by discharges containing metals from the Badger Army Ammunition Plant. The Army agreed to clean up the sediments based on the greater of the CBSQG TEC for mercury or the background concentration, in lieu of doing any additional biological assessments or studies for the site. Since the background concentration for mercury was found to be greater than the TEC value, background was used as the remediation objective. Using CBSQGs to drive cleanup of some sites may be preferable under certain conditions (based on considerations of size of site and defined boundaries of contamination) rather than spending a large amount of time and

resources for additional studies and risk assessments that may lead to considerable costs with little benefit. At larger, more complex sites, the costs associated with detailed studies may be warranted to reduce uncertainties and focus resources on the remedial actions that provide the greatest benefits (MacDonald et al. 1999).

9) It should be noted that there may be contaminated sediment sites and situations where a numerical chemical concentration related to effects may not be the primary driver in a sediment cleanup. Based on a number of balancing factors (e.g., technical feasibility of remediation methods, considerations of natural attenuation factors specific to the site, remedial implementability, human health and ecological risks, stakeholder input, and costs) performance-based standards based on the removal of an established mass of contaminant or removal of visual contamination (applicable to coal tars and petroleum oils) from a site may be the remediation action objective rather than a numerical concentration. There may be situations where the above balancing factors will also be considered to derive a factored cleanup concentration that will not initially achieve the science-based protective sediment concentration but may after an established time period (e.g., when factors such as natural attenuation are considered).

5. Considerations and Advantages of Using Consensus-Based Sediment Quality Guidelines

Given the number of guidelines available, selection of any one as the most appropriate and most reliable for ability to predict toxicity and impacts to benthic species at a study site is difficult. Each guideline set was generally developed using a different methodology (e.g. Ontario [Persaud *et al*. 1993] used the screening level concentration approach and Ingersoll *et al.*[1996a] used the effect level approach). Each approach for developing guidelines has inherent advantages, limitations, levels of acceptance, different extent of field validation, and differing degree of environmental applicability (EPA, 1992). Selecting one set of guidelines is further complicated by uncertainties regarding the bioavailability of contaminants in sediments, the effects of co-varying chemicals and chemical mixtures, the ecological relevance of the guidelines, and correlative versus causal relations between chemistry and biological effects (MacDonald *et al.* 2000a). Given these problems, much discussion has taken place over the use of guidelines as a tool for use in doing sediment quality assessments (Peddicord *et al*. 1998). Cautions are often placed on the use of any one set of guidelines as stand alone decision tools in the assessment and remediation decision making process without additional supporting data from toxicity testing and in-field studies. However, recent evaluations based on combining several sets of guidelines into one to yield "consensus-based" guidelines have shown that such guidelines can substantially increase the reliability, predictive ability, and level of confidence in using and applying the guidelines (Crane *et al*. 2000; MacDonald *et al*. 2000 a, 2000 b; Ingersoll *et al*. 2000). The agreement of guidelines derived from a variety of theoretical and empirical approaches helps to establish the validity of the consensus-based values. Use of values from multiple guidelines that are similar for a contaminant provides a weight-of-evidence for relating to actual biological effects.

A series of papers were produced (Swartz, 1999; Macdonald *et al*. 2000a, 2000b;) that addressed some of the difficulties associated with the assessment of sediment quality conditions using various numerical sediment quality guidelines. The results of these investigations demonstrated that combining and integrating the effect levels from several sets of guidelines to result in consensus-based sediment quality guidelines provide a unifying synthesis of the existing guidelines, reflect causal rather than correlative effects, and can account for the effects of contaminant mixtures in sediment (Swartz, 1999). Additionally, MacDonald et al. (2000a) have evaluated the consensus-based effect levels for reliability in predicting toxicity in sediments by using matching sediment chemistry and toxicity data from field studies conducted throughout the United States. The results of their evaluation showed that most of the consensus-based threshold effect concentrations (TEC - lower effect level) and probable effect concentrations (PEC - upper effect level) for individual contaminants provide an accurate basis for predicting the absence or presence, respectively, of sediment toxicity.

Ingersoll *et al.* (2000, 2001), MacDonald *et al.* (2000a), and Fairey *et al*. (2001) evaluated the reliability of using mean quotient concentration-related values to predict the toxicity in sediments of a mixture of different contaminants. For example, mean PEC quotients were calculated to evaluate the combined effects of multiple contaminants in sediments (Ingersoll et al. 2000, 2001; MacDonald et al. 2000a). A PEC quotient is calculated for each contaminant in each sample by dividing the concentration of a contaminant in sediment by the PEC concentration for that chemical. A mean quotient was calculated for each sample by summing the individual quotient for each contaminant and then dividing this sum by the number of PECs evaluated. Dividing by the number of PEC quotients normalizes the value to provide comparable indices of contamination among samples for which different numbers of contaminants were analyzed. Results of the evaluation showed that the mean PEC quotients that represent mixtures of contaminants were highly correlated to the incidences of toxicity in the same sediments. See Appendix A for calculation methods and ranges of PEC quotient values that are potentially associated with toxicity.

Based on MacDonald *et al*. (2000a), the consensus-based SQGs can be used for or considered for the following:

- To provide a reliable basis for assessing sediment quality conditions in freshwater ecosystems.
- To identify hot spots with respect to sediment contamination.
- To determine the potential for and spatial extent of injury to sediment-dwelling organisms.
- To evaluate the need for sediment remediation.
- To support the development of monitoring programs to further assess the extent of contamination and the effects of contaminated sediment on sediment-dwelling organisms.

The above applications are strengthened when the consensus-based values are used in combination with other sediment quality assessment tools including effects-based testing (i.e., sediment toxicity tests, bioaccumulation assessments, benthic invertebrate community assessments, and more comprehensive designed risk-based studies).

The consensus-based SQGs as developed only involve effects to benthic macroinvertebrate species. The guidelines do not consider the potential for bioaccumulation in aquatic organisms and subsequent food chain transfers to humans or wildlife. Where bioaccumulative compounds are involved, the consensus-based SQGs need to be used in conjunction with other tools, such as bioaccumulation-based guidelines, bioaccumulation studies, food chain modeling, and tissue residue guidelines to evaluate the direct toxicity and upper food chain effects of these compounds.

The MacDonald et al. (2000a) consensus-based sediment quality guidelines have been adopted by the Minnesota Pollution Control Agency (Crane *et al.* 2000) for use as sediment quality targets in the St. Louis River Area of Concern (AOC) on Lake Superior. Following the recommendation in this guidance for the use of the MacDonald *et al.* (2000a) consensus-based SQGs, which would involve their use on the Wisconsin side of the AOC, would be somewhat consistent with their planned use by Minnesota for making assessment and management decisions for contaminated sediment sites on the Duluth side of the AOC.

6. Interpreting Sediment Concentrations That Fall Between the Lower TEC and Upper PEC Consensus-Based Effect Guideline Concentrations

The greatest certainty in predicting the absence or presence of sediment toxicity occurs at sediment contaminant concentrations that are lower than the TEC or greater than the PEC values, respectively. The development of consensus-based SQGs does not include determining the predictability of toxicity related to specific contaminant concentrations in the gradient between the TEC and PEC values. Generally, a consensus-based value for a contaminant cannot be set within the range between the TEC and PEC that would have a low frequency of both false negatives and false positives (Swartz, 1999). Toxicity does occur at contaminant concentrations between the TEC and PEC values with the amount of toxicity dependent on the particular contaminant and with the incidence of toxicity greater than that which occurs at the TEC concentration but less than that which occurs at the PEC concentration (MacDonald *et al.* 2000a). The TEC and PEC concentrations in the consensus-based SQGs define three ranges of concentrations for each contaminant (i.e. < TEC ; > TEC but < PEC ; and > PEC. In assessing the degree of concordance that exists between the chemical concentrations in the three ranges and the incidence of toxicity, it has been demonstrated that for most reliable consensus-based SQG contaminants, there is a consistent and incremental increase in the incidence of toxicity to sediment-dwelling organisms with increasing chemical concentrations (MacDonald *et al*. 2000a, 2000b).

The databases for some individual sets of guidelines, such as the Ontario guidelines (Persaud et al. 1993) that have been combined with other guidelines to produce the consensus-based SQGs can be interpolated to yield predictions of the percent of benthic species that may be affected at specific concentrations between the lower and upper effect levels. A somewhat conservative but still realistic interpretation that can be applied to contaminant concentrations that fall in the gradient of concentrations between the consensus-based TEC and PEC concentrations is that as the concentrations of a contaminant increase, toxicity and effects to benthic macroinvertebrate species related to reductions in survival, reproduction, and growth, bioaccumulation, and benthic community alterations correspondingly increase and/or are increasingly more probable. An identified limitation of this relationship is that the threshold and nature of this trend can be controlled by factors in specific sediments due to their characteristics (Peddicord *et al.*1998). Site specific effects-based testing can be performed to determine the reliability of the prediction of adverse effects based on the use of the CBSQGs on the lower tiers of the assessment.

It is recommended that for the purposes of interpreting the potential impacts of concentrations of contaminants between the TEC and PEC values of the CBSQGs or other guidelines, that a midpoint effect concentration (MEC) be derived and qualitative descriptors be applied to the four possible ranges of concentration that will be created. The qualitative descriptors would be termed "Concern Levels" and would be used as a relative gauge of the potential impacts to the benthic species at that level of contaminant and could be used to prioritize sites for additional studies. A prioritization scheme for ranking sites will, in most cases, depend on professional judgment of staff given the fact that sampling data for sites will generally be variable for the number of samples and the number of parameters analyzed for. The descriptive “Concern Level” scheme is shown in the following table for arsenic concentrations and is applied below in Tables 1 – 4 of the CBSQGs for the various grouped contaminants.

Level of Concern	**Threshold Effect Concentration (TEC)**	**Level of Concern**	**Midpoint Effect Concentration (MEC)**	**Level of Concern**	**Probable Effect Concentration (PEC)**	**Level of Concern**
Level 1	**CBSQG Value**	**Level 2**	**TEC + PEC / 2 = MEC**	**Level 3**	**CBSQG Value**	**Level 4**
≤ TEC		**> TEC ≤ MEC**		**> MEC ≤ PEC**		**> PEC**
Example For CBSQG Values for Arsenic (mg/kg)						
≤ 9.8	9.8	> 9.8 ≤ 21.4	21.4	> 21.4 ≤ 33	33	> 33

7. Recommended Guidelines and Values to be Used in Sediment Quality Assessments

The consensus-based SQG parameters and related effect concentrations in the tables below are from MacDonald *et al.* (2000a) and are indicated in the source column as CBSQGs. Effect-based sediment quality guideline values for some contaminants from other published sources for which CBSQGs were not available are also included in the following tables and identified as such in the source column. These values also represent useful tools for assessing sediment quality. However, their ability to predict toxicity and reliability may not be as great as that for the CBSQGs for a number of reasons including incomplete validation from field testing. This uncertainty has to be weighed in using the values in the assessment process. In cases where more than one set of guidelines have effect-based concentrations for contaminants for which CBSQGs are not available, the effect-based values from that set of guidelines that were the lowest were generally used in the guideline tables that follow. The narrative terminology for effect levels for the latter guidelines may be different from the TEC and PEC terminology from the CBSQGs but the narrative intent is generally the same in establishing a lower and a higher effect level. Also, the emphasis is on those guidelines developed from studies done in freshwater rather than marine or estuarine habitats.

The individual sets of guidelines that were combined and integrated by MacDonald *et al*. (2000a) to yield the CBSQGs are as follows:

Type of SQG	Acronym	Approach	Reference
Derivation of Threshold Effect Concentration (TEC) CBSQG by MacDonald et al. (2000a) from the following			
Lowest Effect Level	LEL	Screening Level Concentration Approach	Persaud *et a*l. 1993
Threshold Effect Level	TEL	Effect Level Approach	Smith *et al.* 1996.
Effect Range - Low	ERL	Effect Level Approach	Long and Morgan, 1991
Threshold Effect Level for *Hyalella azteca* in 28-day tests	TEL-HA28	Effect Level Approach	Ingersoll *et al.* 1996a and 1996b
Minimal Effect Threshold	MET	Screening Level Concentration Approach	EC and MENVIQ, 1992
Chronic Equilibrium Partitioning Threshold	SQAL (Sediment Quality Advisory Level)	Equilibrium Partitioning Approach	Bolton *et al.* (1985); Zarba, (1992); U.S. EPA, 1997
Derivation of Probable Effect Concentration (PEC) CBSQG by MacDonald et al. (2000a) from the following			
Severe Effect level	SEL	Screening Level Concentration Approach	Persaud *et al.* 1993
Probable Effect level	PEL	Effect Level Approach	Smith *et al.* 1996.
Effect Range - Median	ERM	Effect Level Approach	Long and Morgan, 1991
Probable Effect Level for *Hyalella azteca* in 28-day tests	PEL-HA28	Effect Level Approach	Ingersoll *et al.* 1996a and 1996b
Toxic Effect Threshold	TET	Effect Level Approach	EC and MENVIQ, 1992
Acute Equilibrium Partitioning Threshold	No guideline developed	-----	-----

8. Additional Considerations For Some Contaminants

PAHs

Some sources of the parent or unsubstituted PAHs that are in Table 2, such as creosote, coal tars, and petroleum oils, can have co-occurring compounds such as substituted PAHs and heterocyclic aromatic compounds (carbozoles, indoles, acridines, and quinolines) that can be equally or more toxic and more soluble than the listed parent PAH compounds.

Additionally, photoactivation of certain unsubstituted and substituted PAHs, which enhances their toxicity to aquatic organisms that have bioaccumulated these compounds, has been demonstrated both in the laboratory and in the field. The latter may have implications in certain types of habitats (Ankley et al. 2002).

The possible presence of co-occurring toxic compounds where petroleum oils and coal tars are involved and photoactivation of PAHs at sites may need to be considered or toxicity may be underestimated by looking only at the sediment guidelines for the listed parent PAHs in Table 2.

Dioxins and Furans

Polychlorinated dibenzo dioxins (PCDDs) and Polychlorinated dibenzo furans (PCDFs) are unwanted by products of various chemical manufacturing and combustion processes. They are generally ubiquitous in soils and sediments in urban and rural areas. The potential for greatest levels to be found in environmental media are where chlorinated organic compounds such as certain pesticides and pentachlorophenol were either manufactured or used. Pentachlorophenol use at wood treatment operations (railroad ties, utility poles, or lumber) at some sites in Wisconsin sites has led to dioxin and furan compound contamination in floodplain soils and stream sediments. Another source of PCDDs and PCDFs is from the production of paper products from chlorine-bleached wood pulp.

There are 210 polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) which are based on the points of attachment or substitution of chlorine atoms on the aromatic rings. Of these, 17 (7 dioxins and 10 furans) which have chlorine substituted in the 2,3,7,8 positions are thought to pose the greatest risks to receptor organisms. In order to account for the differing toxicities of the 17 2,3,7,8-substituted isomers, each has been given a toxic equivalency factor (TEF) related to the most toxic form, 2,3,7,8-TCDD (TEF = 1.0). In terms of risk assessments, those PCDDs and PCDFs not substituted in the 2,3,7,8 positions can be ignored. The summed concentration of the TEF of each 2,3,7,8-substituted isomer times its concentration equals the toxic equivalent concentration to 2,3,7,8-TCDD or TCDD-EQ concentration. Appendix C provides a table to calculate a summed TCDD-EQ concentration based on the TEF value and reported concentration for each of the 17 2,3,7,8-substituted isomers found in sediments and floodplain soils.

Cyanide

Cyanide as measured and reported as total cyanides in sediments can include hydrogen cyanide (HCN), cyanide ion (CN^-), simple cyanides, and metallo- and organo-cyanide complexes. HCN and CN^- are grouped as free cyanides and are the most toxic forms of cyanide and the forms of concern.

Most complexed cyanides are relatively nontoxic and total cyanide determinations are not very useful measures of either water or sediment quality. Factors that affect the release or dissociation of free cyanides from complexed cyanide forms include pH, redox potential, photodecomposition of the complex and release of free cyanide, relative strength of the metallo- and organo-cyanide complexes, and possible presence of bacteria responsible for degradation of ferrocyanide complexes. In sediments, the cyanide in the free form present in the pore water is more relatable to toxicity to benthic organisms than the total cyanide measured in the solid phase. However, given the above factors, it is difficult to predict or model the dissociation and release of the free toxic forms of cyanide to the pore water from the less toxic total cyanide form associated with and normally measured in the solid phase sediments. A general idea of the concentrations of free cyanide in pore water that would be toxic to benthic invertebrates can be drawn from the acute and chronic toxicity criteria for free cyanides in surface waters classified as supporting Warm Water Sport Fish (NR 105, Wis. Admin. Code) which are 45.8 ug/L and 11.47 ug/L, respectively. Free cyanides as HCN, in general, are not very persistent in the environment due to their volatility, have low adsorption to sediment particles, high water solubility, and inability to substantially bioaccumulate. Where any significant levels of total cyanide are detected in sediments, additional analysis may need to be done to also determine what fractions of the total cyanide are in dissociable forms (amenable to chlorination or weak acid dissociable forms) to give an indication of the potential to release free cyanide with its attendant toxicity..

9. Background or Reference Site Concentration Considerations In Using the Effect-Based SQGs

In designing and collecting sediment samples at any phase of a site assessment, consideration may need to be given to sampling and analyzing for the same potential chemical stressors, biological data, and/or physical data that are being analyzed for within the study site area at a representative background/reference site to be used as benchmarks for comparison purposes. Establishing representative reference sites is critical because if reference sites are not highly similar to the areas under study, misleading or inappropriate conclusions may be drawn when making data comparisons (Apitz *et al.* 2002). The background/reference site selected needs to have all the characteristics of the study site sediments as close as practical, which includes similar particle size fractions, total organic carbon content, depositional attributes, and relative positioning (e.g., water depth and stream cross section) in the water body as the study site location, but needs to be out of the influence of the study site and the factors responsible for contaminating the study site. Contributions of contaminants (see Appendix E for a discussion of contamination/contaminant and relation to adverse effects) at the reference site can come from two sources: 1) natural sources based on the soils and geological features in the watershed, and 2) anthropogenic sources such as urban runoff. The reference site should be relatively unaffected by anthropogenic inputs. In urban areas, sediment sites outside of the factors that may be influencing the study site may themselves be influenced by ubiquitous urban sources. The sediment quality of reference sites should be reflective of the land uses and land cover of the watershed that the study site is in. Alternatively, suitable background values may be derived through sediment profiles by examining concentrations at depth with the assumption that the lowest concentration at depth represents the pre-industrial or pre-development sediment horizon (Persaud *et al*. 1993).

It has to be recognized that in diverse geographical and geological areas, the natural levels of metals and ubiquitous source anthropogenic organic compounds will vary. Given this variation, dependence should be put on site-specific samples for establishing reference site concentrations rather than depending on data compiled from other unrelated sites. In areas and at sites where the background/reference site concentrations are greater than the CBSQG TEC values, the local background/reference site concentrations should be used as the practical lower limit for doing sediment evaluations and making management decisions for additional sediment assessments.

The particle size fractions (for metals) and total organic carbon (TOC) content (for nonpolar organic compounds) of all samples should be used to normalize concentrations in order to do relevant and appropriate site-to-site comparisons of contaminant concentrations.

TOC can have its origin either from organic matter from natural sources such as plant materials deposited on sediments or anthropogenic inputs to aquatic systems. In the latter case, elevated TOC sources in sediments can be from such sources as residual petroleum oils, coal tars, or creosote. The controlling importance of the amount of natural organic matter as a TOC source for determining the fate and bioavailability of organic chemicals, especially nonpolar or neutral compounds, has been established (U.S. EPA, 1993). A chemically-unique partitioning coefficient (K_{OC}) for a nonpolar organic compound is used to estimate the pore water concentration based on its partitioning from natural TOC in the sediment. The partitioning coefficient for a compound is assumed to be relatively constant and predictable across various types of natural organic matter. The K_{OC} values for organic compounds can be found in chemical reference books. Nonpolar organic compounds associated with residual oils of anthropogenic origin as a partition media will have different partitioning coefficients compared to natural organic matter (Boyd and Sun, 1990 and Sun and Boyd, 1991) due to the quality of organic carbon. The latter situation may need to be addressed when estimating the bioavailability of nonpolar organic compounds where the TOC is predominantly contributed by some sources of anthropogenic origin.

For metals and particle size, comparing the concentrations of a contaminant in a sample dominated by a fine fraction with one dominated by a sand fraction would be inappropriate and would not yield useful information. Metals and anthropogenic organic compounds will tend to sorb and concentrate in or on finer grained sediments and TOC, respectively.

The intensity of sampling for establishing representative background/reference site concentrations of contaminants should increase at upper tiers in the sediment evaluation process. For example, for comparisons done in the lower tiers of an assessment when initially investigating the site, one to three sediment samples from the reference site, either analyzed individually or composited for one analysis may be appropriate. Where the reference site concentration comparisons may play a more important role in evaluation and management decisions for a site at upper tiers of an assessment, the sampling intensity should generally increase, with at least 10 or more samples taken at the reference site and analyzed individually. Data sets with fewer than 10 samples generally provide for poor estimates of mean concentrations (i.e., there is a large difference between the sample mean and the 95% upper confidence limit). In most cases, a maximum probable background concentration (MPBC) should be calculated for the contaminant(s) derived from the upper 95% confidence level of the mean (EPA, 1992b) after consideration of the distribution of the sample concentrations as showing either a normal or log normal distribution (see Appendix B for example calculations).

Sample results for a metal or organic compound of concern at the background/reference site may be reported out as a censored value i.e. less than a detection level based on the analytical method that meets the data quality objectives established for the sampling and analysis. There are various methods to handle the censored data to derive values that can be used with the uncensored values in the data set to derive a mean and standard deviation to be used in the calculation of a maximum probable background concentration. Analyses of methods to handle censored data show that, in most cases, sophisticated statistical techniques recommended for estimation problems involving censored data are unnecessary or even inappropriate for statistical comparisons where the number of censored data samples in a data set are generally small. In general, the simple substitution methods work best to maintain power and control type I error rate in statistical comparisons (Clarke, 1995). The simple substitution method includes either 1) substitution of the detection limit as the quantified concentration, or 2) substitution of one-half the detection limit as the quantified concentration. Clarke (1995) recommends steps in selecting the substitution method. At its simplest, substitution method 1) above should generally be used where the number of censored data results are less than 40% of the data set, and method 2) where the censored data is greater than 40%.

9.1 Metals and Silt/Clay Fraction Relationships

There is a strong correlation between decreasing grain size and increasing metal concentrations. Sand-sized material, which is typically low in trace metal concentrations, may serve as a diluent of metal-rich finer grained particles. Larger fractions of sand can hide significant trace metal concentrations and dispersion patterns (Horowitz, 1991). Adjusting for particle grain size effects is important for 1) determining natural background levels of trace elements associated with sediments to serve as a baseline for comparison purposes with other sites, 2) for distinguishing and determining the degree of anthropogenic enrichment, 3) for comparing metal data from site-to-site on a standardized basis, and 4) providing a means for tracing the extent of metal transport and dispersion by eliminating the diluent effects of large particle size contributions.

Two methods are used to address grain size effects. One is to separate out the sand, silt, and clay sized particles from a sample by sieving and analyzing the separate fractions. The other method is to assume that the majority of the metals in a sample are associated with the fine fraction (silt + clay) and then mathematically normalize the metal data to this fraction by dividing the bulk concentration by the fine fraction percentage expressed as a decimal fraction to yield mg of a metal / kg of fines. Particle size analysis of a sediment sample is usually reported as percent sand, silt, and clay fractions. An example of normalizing a bulk sediment concentration for a metal to the fine fraction for a sample with 84 mg/kg of lead and 60% fines (40% silt + 20% clay) is 84 mg Pb/kg ÷ 0.60 kg fines /kg sediment = 140 mg lead / kg of fines. The assumption may not always hold true that all or most of the metals are associated with the fine fraction. Also, when the fine fraction falls below 50% of the total combined fractions, the mathematical normalization may not represent the true metal concentration in the fines (Horowitz, 1991). The normalization to the fine fractions should at a minimum be done at least qualitatively to compare on a relative basis the fine fraction contents between the sediment samples where the metal concentrations are being compared. Besides grain size, other normalizing factors have been used and include iron, aluminum, and total organic carbon (Daskalakis *et al.* 1995).

It should be noted that for the CBSQGs for the metals, MacDonald et al. (2000a) do not indicate what the relative percentage of the mineral particle size fractions (% sand, silt, and clay) were assumed to be associated with the expressed values. TOC may play some role in the chemical form of the metal and thus its release from the sediments and its bioavailability. TOC may serve as a secondary binding phase of metals with acid volatile sulfates (AVS) serving as the primary binding phase. It is difficult to predict or measure the role of TOC as it relates to metals. For this reason, the study site bulk sediment metal concentrations need to be directly compared with the CBSQG concentrations in Table 1 without any adjustments for TOC or fine fraction content. The process above for adjusting metal concentrations based on the percent fines is an additional assessment tool for comparing the concentrations between the unimpacted reference site and the study site and between study sites on a fine content-normalized basis and does not play a role in SQG application.

Normalizing contaminant concentrations to the mineral fine content or TOC content is not to be done for assessing toxicity under TSCA or determining hazardous waste characteristics under the Toxicity Characteristic Leaching Procedure (TCLP) test. The sample dry weight bulk concentrations as reported by the analytical laboratory are to be used for comparison with the applicable criteria under these regulations.

9.2 Nonpolar Organic Compound and Total Organic Carbon Relationships

In the case of nonpolar organic compounds such as PAHs, PCBs, dioxins/furans, and chlorinated pesticides, the bulk sediment concentrations can be normalized to the TOC content for site-to-site comparison purposes by dividing the dry weight sediment concentration by the percent TOC in the sediment expressed as a decimal fraction. For example the TOC normalized PCB concentration for a sediment concentration of 7 mg/kg with 3.5% TOC is 200 mg PCB / kg TOC (i.e., 7 mg PCBs/kg ÷ 0.035 kg TOC/kg = 200 mg PCB/kg TOC). Normalization of nonpolar organic compounds to TOC content is valid only if the TOC content in the sediments is greater than 0.2%. At TOC concentrations less than 0.2%, other factors that influence partitioning to the sediment pore waters (e.g., particle size and sorption to nonorganic mineral fractions) become relatively more important (Di Toro *et al.*1991).

MacDonald *et al.* (2000a) indicate that some individual sets of guidelines that were used in their consensus-based approach were originally expressed on an organic carbon-normalized basis. They converted the values in these sets of to dry weight-normalized values at 1% organic carbon to be averaged with the other sets of guideline values to yield the CBSQGs. The final MacDonald *et al.* (2000a) CBSQG values are expressed on a dry weight basis without regard to organic carbon content. It should be noted that the consensus-based SQG values in Tables 2, 3, and 4 below are expressed on an assumed <u>dry weight normalized basis at 1% organic carbon</u>. It has been established that the organic carbon content of sediment is an important factor influencing the movement and bioavailability of nonpolar organic compounds (e.g., PAHs, PCBs, and chlorinated pesticides) between the organic carbon content in bulk sediments and the sediment pore water and overlying surface water. Biological responses of benthic organisms to nonionic organic chemical in sediments are different across sediments when the sediment concentrations are expressed on a dry weight basis, but similar when expressed on an organic carbon normalized basis (ug chemical / g organic carbon basis) (U.S. EPA, 2000).

To appropriately compare the CBSQG dry weight-normalized to 1% TOC values with the dry weight concentrations in the study sediments of variable TOC content, the study sediment contaminant concentrations also need to be converted to a dry weight-normalized to 1% TOC basis.
Appendix D provides a spread sheet for calculating dry weight sediment concentrations for nonpolar organic compounds normalized to 1% TOC. The concentrations given are for an example sediment. Appendix D also contains a spreadsheet for calculating the concentrations of metals normalized to the fine fraction in a sediment sample. An Excel spreadsheet is available for doing the calculations.

An example showing the necessity of doing this conversion to a common 1% TOC basis for organic compounds is shown as follows:

- The threshold effect concentration (TEC) for total PAHs (TPAHs) is 1,610 ug/kg at 1% TOC.
- The example site under assessment has a TPAH concentration of 7,300 ug/kg at 5% TOC.
- Comparing the dry weight concentrations between the guideline value and the example site concentration without consideration of the TOC content differences would appear to show that the study site concentrations are greater than the TEC guideline value (7,300 study site vs. 1,610 TEC).
- To convert the study site TPAH concentration to a dry weight concentration normalized to 1%, divide the 7,300 ug/kg value by 5 (5% TOC content) = 1,460 ug TPAH/kg at 1% TOC. On the common basis of 1% TOC, the study site TPAH concentration is less than the TEC concentration (1,460 ug/kg study site vs. 1,610 ug/kg TEC).
- In the case above, another approach for converting the concentrations to a common normalized basis is to multiply the TEC concentration by 5 that is the percent TOC of the study site sample. The common basis here are dry weight-normalized concentrations at 5% TOC (7,300 ug/kg study site vs. 8,050 ug/kg TEC).

10. Point of Application of the CBSQGs in the Bed Sediment

The numerical CBSQGs apply to the biologically active zone associated with deposited sediments in flowing (streams and rivers) and static (lakes and ponds) water bodies and wetland soils and sediments. The biologically active zone is inhabited by infaunal organisms including microbes, meiofauna, and macroinvertebrates and other organisms (e.g., egg and larval stage of fish) that spend all or part of their life cycles associated either within (infaunal) or on (epibenthic) the bottom sediments. The community of organisms present will generally depend on the physical and chemical characteristics of the waterbody and bottom sediments as determined by the watershed location and ecoregion within the State. The depth of the biologically-active zone varies between sites depending on the substrate characteristics present (including particle size fractions, organic matter content, compaction, pore-water geochemistry, and water content) which influence the composition of sediment-associated organisms present. The biologically active zone typically encompasses the top 20 to 40 cm. of sediment in freshwater environments (Clarke et al. 2001). The majority of benthic organisms will usually be associated with the upper strata (e.g., 15 cm) related to these depth ranges. Certain invertebrate and/or amphibian species can utilize habitats deeper in bed sediments during a portion of their life history (e.g., down to 100 cm below the sediment surface) (MacDonald *et al*. 2000a). The best available knowledge about the local composition of sediment-associated biota and the bioactive depth zone they occupy should supplement the generic depth assumptions above (Clarke *et al*. 2001) where possible. Contaminants in sediments at depths below the biologically active zone can be of concern because of their potential to move to the upper sediment strata through various mechanisms that include diffusion and being transported on groundwater flows that discharge to the surface water body. The groundwater-sediment-surface water zone is a zone of transitions in which various environmental factors can affect contaminant fate and transport.

The CBSQGs should be considered when assessing contaminated soils and sediments deposited on upper bank areas and floodplain areas that have the potential to be eroded or scoured and transported to and deposited in a nearby surface water body.

11. Other Approaches Being Used to Develop SQGs

U.S. EPA has developed national equilibrium partitioning sediment guidelines (ESGs) for a broad range of sediment types. They have finalized the methodologies for deriving ESGs for nonionic organic chemicals (2000a) and mixtures of certain metals (cadmium, copper, lead, nickel, zinc, and silver (U.S.EPA, 2000b). U.S. EPA is planning to publish final guidance (EPA, 2000c) for developing SQGs based on a combination of the equilibrium partitioning (EqP) approach, quantitative structure activity relationships, narcosis theory, and concentration addition models for mixtures of PAH found at specific sites. The EqP-based summed PAH toxicity model provides a method to address causality, account for bioavaliability, consider mixtures, and predict toxicity and ecological effects (U.S. EPA, 2000). The U.S. EPA guidance indicates that the total number of PAHs that need to be considered in SQG development is 34 (18 parent and 16 with alkylated groups). Use of fewer than 34 may greatly underestimate the total toxicological contribution of PAH mixtures. The guidance requires the use of conservative uncertainty factors to be applied when fewer than the 34 are being used to estimate site-specific toxicity of PAH mixtures.

When guidance has been published in final for the use and application of the ESGs for metals, PAH mixtures, and other nonionic organic compounds, the Water Quality Standards section plans to produce additional guidance on the use of the ESGs to be used in addition to or instead of the CBSQGs. U.S. EPA's apparent intent is not to use the ESG numeric values as stand alone criteria for application as part of a States water quality standards under Section 3 (c) of the Clean Water Act, but to use them as a screening tool in conjunction with other assessment tools such as toxicity testing in evaluating and prioritizing sites under various programs (e.g., developing Total Maximum Daily Loads (TMDLs) s and WPDES permit limitations, Superfund, RCRA).

Table 1. Recommended Sediment Quality Guideline Values For Metals and Associated Levels of Concern To Be Used In Doing Assessments of Sediment Quality.

Metal	mg/kg dry wt.[++]							
	Level 1 Concern ≤ TEC	TEC	Level 2 Concern > TEC ≤ MEC	MEC	Level 3 Concern > MEC ≤ PEC	PEC	Level 4 Concern > PEC	Source of SQG Effect-Based Concentrations
Antimony	⇦	**2**	⇔	**13.5**	⇔	**25**	⇨	NOAA (1991) [1.]
Arsenic	⇦	**9.8**	⇔	**21.4**	⇔	**33**	⇨	CBSQG (2000a)[2]
Cadmium	⇦	**0.99**	⇔	**3.0**	⇔	**5.0**	⇨	CBSQG (2000a)
Chromium	⇦	**43**	⇔	**76.5**	⇔	**110**	⇨	CBSQG (2000a)
Copper	⇦	**32**	⇔	**91**	⇔	**150**	⇨	CBSQG (2000a)
Iron	⇦	**20,000**	⇔	**30,000**	⇔	**40,000**	⇨	Ontario (1993) [3]
Lead	⇦	**36**	⇔	**83**	⇔	**130**	⇨	CBSQG (2000a)
Manganese	⇦	**460**	⇔	**780**	⇔	**1,100**	⇨	Ontario (1993)
Mercury	⇦	**0.18**	⇔	**0.64**	⇔	**1.1**	⇨	CBSQG (2000a)
Nickel	⇦	**23**	⇔	**36**	⇔	**49**	⇨	CBSQG (2000a)
Silver	⇦	**1.6**	⇔	**1.9**	⇔	**2.2**	⇨	BC (1999) [4.]
Zinc	⇦	**120**	⇔	**290**	⇔	**460**	⇨	CBSQG (2000a)

++ The CBSQGs for organic compounds are expressed on a dry weight concentration at 1% TOC in sediments. However, unlike the organic compounds, the CBSQG and study site metals concentrations can be compared on a bulk chemistry basis and do not need to be adjusted to a 1% TOC basis to do the comparison. TOC does not play the same role in determining metals availability as it does in determining organic compound availability.

1. NOAA (1991) = Long, E.R. and L.G. Morgan. 1991. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52. National Oceanic and Atmospheric Administration. Seattle, Washington.

2. CBSQG (2000a) = MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000a. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Arch. Environ. Contam. Toxicol. 39:20-31.

3. Ontario (1993) = Persaud, D.R., R. Jaagumagi, and A. Hayton. 1993. Guidelines for the protection and management of aquatic sediments in Ontario. Standards Development Branch. Ontario Ministry of Environment and Energy. Toronto, Canada.

4. MacDonald, D.D. and M. MacFarlane. 1999. (Draft). Criteria for managing contaminated sediment in British Columbia. British Columbia Ministry of Environment, Lands, and Parks. Victoria, British Columbia.

Table 2. Recommended Sediment Quality Guideline Values For Polycyclic Aromatic Hydrocarbons (PAHs) and Associated Levels of Concern To Be Used In Doing Assessments of Sediment Quality.

PAH	ug/kg dry wt. at 1% TOC ++							
	Level 1 Concern ≤ TEC	TEC	Level 2 Concern > TEC ≤ MEC	MEC	Level 3 Concern > MEC ≤ PEC	PEC	Level 4 Concern > PEC	Source of SQG Effect-Based Concentrations
Low Molecular Weight PAHs (3 or less benzene rings)								
Acenapthene	⇦	6.7	⇔	48	⇔	89	⇨	CCME (1999) [1.]
Acenaphthylene	⇦	5.9	⇔	67	⇔	128	⇨	CCME (1999)
Anthracene	⇦	57.2	⇔	451	⇔	845	⇨	CBSQG (2000a) [2.]
Fluorene	⇦	77.4	⇔	307	⇔	536	⇨	CBSQG (2000a)
Naphthalene	⇦	176	⇔	369	⇔	561	⇨	CBSQG (2000a)
2-methylnapthalene	⇦	20.2	⇔	111	⇔	201	⇨	CCME (1999)
Phenanthrene	⇦	204	⇔	687	⇔	1,170	⇨	CBSQG (2000a)
High Molecular Weight PAHs (4 or more benzene rings)								
Benz(a)anthracene	⇦	108	⇔	579	⇔	1,050	⇨	CBSQG (2000a)
Benzo(a)pyrene	⇦	150	⇔	800	⇔	1,450	⇨	CBSQG (2000a)
Benzo(e)pyrene	⇦	150	⇔	800	⇔	1,450	⇨	Similar as above [3.]
Benzo(b)fluoranthene	⇦	240	⇔	6,820	⇔	13,400	⇨	Similar as below [4.]
Benzo(k)fluoranthene	⇦	240	⇔	6,820	⇔	13,400	⇨	Persaud *et al*. 1993 [5]
Benzo(g,h,l)perylene	⇦	170	⇔	1,685	⇔	3,200	⇨	Persaud *et al.* 1993
Chrysene	⇦	166	⇔	728	⇔	1,290	⇨	CBSQG (2000a)
Dibenz(a,h)anthracene	⇦	33	⇔	84	⇔	135	⇨	CBSQG (2000a)
Fluoranthene	⇦	423	⇔	1,327	⇔	2,230	⇨	CBSQG (2000a)
Indeno(1,2,3-cd)pyrene	⇦	200	⇔	1,700	⇔	3,200	⇨	CBSQG (2000a)
Pyrene	⇦	195	⇔	858	⇔	1,520	⇨	CBSQG (2000a)
Total PAHs								
Total PAHs	⇦	**1,610**	⇔	**12,205**	⇔	**22,800**	⇨	CBSQG (2000a)

++ To compare the study site concentrations with the Table 2 concentrations on a common basis, divide the study site concentrations by the %TOC at the study site to yield a dry wt. normalized value at 1% TOC. If no site TOC information is available, assume a 1% TOC content.

1. CCME (1999) = Canadian Council of Ministers of the Environment (CCME). 1999. Canadian sediment quality guidelines for the protection of aquatic life: Summary tables. In: Canadian environmental quality guidelines. 1999. Canadian Council of Ministers of the Environment, Winnipeg.
2. CBSQG (2000a) = MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000a. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Arch. Environ. Contam. Toxicol. 39:20-31.
3. There are no guideline values for Benzo(e)pyrene. "Similar as above" assumes the similarity of the chemical structure of Benzo(e)pyrene with Benzo(a)pyrene would yield similar quantitative structure activity relationships (QSARs) as it relates to toxicity, therefore the effect level concentrations that were derived for Benzo(a)pyrene would also apply to Benzo(e)pyrene.
4. There are no guideline values for Benzo(b)fluoranthene. "Similar as below" assumes the similarity of the chemical structure of Benzo(b)fluoranthene with Benzo(k)fluoranthene would yield similar quantitative structure activity relationships (QSARs) as it Relates to toxicity, therefore the effect level concentrations that were derived for Benzo(k)fluoranthene would also apply to Benzo(b)fluoranthene.
5. Ontario (1993) = Persaud, D.R., R. Jaagumagi, and A. Hayton. 1993. Guidelines for the protection and management of aquatic sediments in Ontario. Standards Development Branch. Ontario Ministry of Environment and Energy. Toronto, Canada.

Table 3. Recommended Sediment Quality Guideline Values For Polychlorinated Biphenyls (PCBs) And Chlorinated and Other Pesticides and Associated Levels of Concern To Be Used In Doing Assessments of Sediment Quality.

PCB and Pesticides	ug/kg dry wt. at 1% TOC ++							
	Level 1 Concern ≤ TEC	TEC	Level 2 Concern > TEC ≤ MEC	MEC	Level 3 Concern > MEC ≤ PEC	PEC	Level 4 Concern > PEC	Source of SQG Effect-Based Concentrations
PCBs								
Total PCBs	⇦	60	⇔	368	⇔	676	⇨	CBSQG (2000a) [1.]
Pesticides								
Aldrin	⇦	2	⇔	41	⇔	80	⇨	Ontario (1993) [2.]
BHC	⇦	3	⇔	62	⇔	120	⇨	Ontario (1993)
alpha-BHC	⇦	6	⇔	53	⇔	100	⇨	Ontario (1993)
beta-BHC	⇦	5	⇔	108	⇔	210	⇨	Ontario (1993)
gamma-BHC (lindane)	⇦	3	⇔	4	⇔	5	⇨	CBSQG (2000a)
Chlordane	⇦	3.2	⇔	10.6	⇔	18	⇨	CBSQG (2000a)
Dieldrin	⇦	1.9	⇔	32	⇔	62	⇨	CBSQG (2000a)
Sum DDD	⇦	4.9	⇔	16.5	⇔	28	⇨	CBSQG (2000a)
Sum DDE	⇦	3.2	⇔	17	⇔	31	⇨	CBSQG (2000a)
Sum o,p' + p,p' DDT	⇦	4.2	⇔	33.6	⇔	63	⇨	CBSQG (2000a)
Sum of DDT +DDD + DDE	⇦	5.3	⇔	289	⇔	572	⇨	CBSQG (2000a)
Endrin	⇦	2.2	⇔	653	⇔	207	⇨	CBSQG (2000a)
Heptachlor Epoxide	⇦	2.5	⇔	9.3	⇔	16	⇨	CBSQG (2000a)
Mirex	⇦	7	⇔	10.5	⇔	14	⇨	BC (1999) [3.]
Toxaphene	⇦	1	⇔	1.5	⇔	2	⇨	BC (1999)

++ To compare the study site concentrations with the Table 3 concentrations on a common basis, divide the study site concentrations by the %TOC at the study site to yield a dry wt. - normalized value at 1% TOC. If no site TOC information is available, assume a 1% TOC content.

1. CBSQG (2000a) = MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000a. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Arch. Environ. Contam. Toxicol. 39:20-31.

2. Ontario (1993) = Persaud, D.R., R. Jaagumagi, and A. Hayton. 1993. Guidelines for the protection and management of aquatic sediments in Ontario. Standards Development Branch. Ontario Ministry of Environment and Energy. Toronto, Canada.

3. MacDonald, D.D. and M. MacFarlane. 1999. (Draft). Criteria for managing contaminated sediment in British Columbia. British Columbia Ministry of Environment, Lands, and Parks. Victoria, British Columbia.

Table 4. Recommended Sediment Quality Guideline Values For Assorted Contaminants and Associated Levels of Concern To Be Used In Doing Assessments of Sediment Quality.

Sediment Contaminant	ug/kg dry wt. at 1% TOC ++							
	Level 1 Concern ≤ TEC	TEC	Level 2 Concern > TEC ≤ MEC	MEC	Level 3 Concern > MEC ≤ PEC	PEC	Level 4 Concern > PEC	Source of SQG Effect-Based Concentrations
Benzene	⇦	**57**	⇔	**83.5**	⇔	**110**	⇨	BC (1999) [1.]
Toluene	⇦	**890**	⇔	**1,345**	⇔	**1,800**	⇨	BC (1999)
Xylene	⇦	**25**	⇔	**37.5**	⇔	**50**	⇨	BC (1999)
2,3,7,8-TCDD (pgTEQ/g)	⇦	**0.85**	⇔	**11.2**	⇔	**21.5**	⇨	Canada (2002) [2.]
Pentachlorophenol	⇦	**150**	⇔	**175**	⇔	**200**	⇨	Janisch (1990) [3.]
Tributyltin	⇦	**0.52**	⇔	**1.73**	⇔	**2.94**	⇨	Janisch (1994) [4.]
1,2-Dichlorobenzene	⇦	**23**	⇔	**-----**	⇔	**23**	⇨	Washington (1991) [5.]
1,4-Dichlorebenzene	⇦	**31**	⇔	**60.5**	⇔	**90**	⇨	Washington (1991)
1,2,4-Trichlorobenzene	⇦	**8**	⇔	**13**	⇔	**18**	⇨	Washington (1991)
Dimethyl Phthalate	⇦	**530**	⇔	**-----**	⇔	**530**	⇨	Washington (1991)
Diethyl Phthalate	⇦	**610**	⇔	**855**	⇔	**1,100**	⇨	Washington (1991)
Di-N-Butyl Phthalate	⇦	**2,200**	⇔	**9,600**	⇔	**17,000**	⇨	Washington (1991)
Di-N-Octyl Phthalate	⇦	**580**	⇔	**22,790**	⇔	**45,000**	⇨	Washington (1991)
Dibenzofuran	⇦	**150**	⇔	**365**	⇔	**580**	⇨	Washington (1991)
Phenol	⇦	**4,200**	⇔	**8,100**	⇔	**12,000**	⇨	Washington (1991)
2-Methylphenol	⇦	**6,700**	⇔	**-----**	⇔	**6,700**	⇨	Washington (1991)
2,4-Dimethyl Phenol	⇦	**290**	⇔	**-----**	⇔	**290**	⇨	Washington (1991)
Benzyl Alcohol	⇦	**570**	⇔	**650**	⇔	**730**	⇨	Washington (1991)
Benzoic Acid	⇦	**6,500**	⇔	**-----**	⇔	**6,500**	⇨	Washington (1991)

++ To compare the study site concentrations with the Table 4 concentrations on a common basis, divide the study site concentrations by the %TOC at the study site to yield a dry wt. - normalized value at 1% TOC. If no site TOC informatio is available, assume a 1% TOC content.

1. MacDonald, D.D. and M. MacFarlane. 1999. (Draft). Criteria for managing contaminated sediment in British Columbia. British Columbia Ministry of Environment, Lands, and Parks. Victoria, British Columbia.
2. Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Summary Table. Update 2002. Canadian Council of Ministers of the Environment.
3. Janisch (1990) = Memo of February 7, 1990 prepared to Maltbey of NCD entitled Sediment Quality Criteria for Pentachlorophenol related to the Semling-Menke Company Contaminated Groundwater Inflow to the Wisconsin River. Sediment guidelines for Developed for pentachlorophenol in sediment based on the water quality criteria in NR 105. Considerations made for pH of of water and organic carbon partitioning coefficient of pentachlorophenol. The pH determines the dissociated / undissociated forms of pentachlorophenol and its partitioning coefficient. The pH used to calculate the above sediment values was 7.0. The K_{oc} value used was 3.226 or 1,821 L/kg OC. The organic carbon content of the sediment was assumed to be 1%. The TEC and PEC values above for PCP were based on the chronic and acute water quality criteria in NR 105, respectively.
4. Janisch (1994) = Memo of November 14, 1994 prepared to LaValley of NWD entitled Preliminary Ecological Risk Assessment for the Contaminated Sediments Associated with the Fraser Shipyard Site, Superior, Wisconsin. Sediment guidelines for tributyltin derived based on the proposed water quality criteria for tributyltin at the time (EPA, 1988). The organic carbon partitioning coefficient used was 1,970 L/kg OC and an assumed organic carbon content of 1% in sediment. The TEC and PEC values above for tributyltin were based on the chronic and acute water quality values as proposed by EPA, respectively.
5. Washington (1991) = Sediment Management Standards, Chapter 173-204 WAC, Washington State Department of Ecology. April 1991. The Standards were developed using the Apparent Effects Threshold Approach. The TEC and PEC values above for the compounds are based on no effect and minimal effect standards, respectively, from the Washington Standards and are intended to apply to Puget Sound, an estuarine habitat. The values were calculated based on an assumed TOC content in sediment of 1%.

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Appendix A

Recommended Procedure for Calculating Mean Probable Effect Quotients (Mean PEC Quotients) for Mixtures of Chemicals found at Contaminated Sediment Sites and Their Reliability of Predicting the Presence or Absence of Toxicity (Adopted from Ingersoll *et al.* 2000, 2001).

Step 1. Based on existing databases, the reliability to predict toxicity is greatest for the organic compound groups of total PAHs and total PCBs and the metals arsenic, cadmium, chromium, copper, lead, nickel, and zinc. Inclusion of other compounds or metals that have a PEC value, where there is insufficient data available to evaluate its predictive reliability (e.g., mercury, dieldrin, DDD, DDT, endrin, and lindane) into the overall PEC-Q calculation may result in an overall PEC-Q value with lower predictive ability.

Step 2. Calculate the individual PEC Quotients (PEC-Qs) for chemicals with reliable PECs within each of the chemical classes. Since the PECs for PAH and PCB chemical classes are based on total concentrations, individual PEC-Qs for individual compounds in these classes do not need to be calculated.

$$\text{Individual Chemical PEC-Q} = \frac{\text{Chemical concentration in Study Site Sediments (in dry wt.)}}{\text{PEC SQG Concentration for Chemical (in dry wt.)}}$$

For the nonpolar organic compounds (total PCBs and total PAHs), the PEC SQG is expressed on a dry weight basis normalized to 1% organic carbon. The concentration for these groups of nonpolar compounds in the study site sediments also needs to be expressed on this same basis. To do this, divide the concentration in the study site sediments by the percent TOC in the sediments expressed as a whole number (e.g., 7,300 ug/kg PCB at 5% TOC is 7,300 ÷ 5 = 1,460 mg/kg dry weight normalized to 1% TOC).

Step 3. In the case of metals, a mean $\text{PEC-Q}_{\text{metals}}$ for the metals involved needs to be calculated based on summing the PEC-Q for the individual metals and dividing by the number of metals.

$$\text{Mean PEC-Q}_{\text{metals}} = \frac{\Sigma \text{ individual metal PEC-Qs}}{\text{Number of metals for which individual PEC-Qs calculated}}$$

Step 4. Calculate the overall mean PEC-Q for the three main classes of chemicals.

$$\text{Mean PEC-Q}_{\text{overall}} = \frac{(\text{mean PEC-Q}_{\text{metals}} + \text{PEC-Q}_{\text{total PAHs}} + \text{PEC-Q}_{\text{total PCBs}})}{n}$$

Where n = number of classes of chemicals for which sediment chemistry available (e.g., in this case, there are three classes – metals, PAHs and PCBs. In other cases, metals and PAHs may be the only chemicals of concern at a site and therefore PEC-Qs may only be calculated for these two groups and therefore n = 2.

Appendix A (continued)

The database used by Ingersoll et al. (2001) to determine the ability of the PEC-Qs to predict toxicity is based on testing freshwater sediments from a number of sites using 10- to 42-day toxicity tests with the amphipod *Hyalella azteca* or the 10- to 14-day toxicity tests with the midges *Chironomus tentans* or *C. riparius.* Toxicity of samples was determined as a significant reduction in survival or growth of the test organisms relative to a control or reference sediment. A relative idea of the predictive ability of the overall mean PEC-Qs and individual PEC-Qs for each group of chemicals is shown in the table below from Ingersoll et al. (2001). Mean PEC quotients were calculated to provide an overall measure of chemical contamination and to support an evaluation of the combined effects of multiple contaminants in sediments.

Test Species and Test Duration	**Incidence of Toxicity (% of samples where toxicity observed versus no toxicity) Based on the Mean PEC Quotients (Number of Samples in Parentheses)**					**Total Number of Samples**
	Range of Mean PEC Quotients					
	< 0.1	**0.1 to < 0.5**	**0.5 to < 1.0**	**1.0 to < 5.0**	**> 5.0**	
Hyalella azteca 10- to 14-day tests						
Mean Overall PEC-Q [1.]	**19** (79)	**26** (89)	**38** (34)	**49** (35)	**86** (29)	266
Q$_{metals}$ [2.]	**23** (40)	**24** (139)	**33** (45)	**81** (31)	**100** (11)	266
PEC-Q$_{total\ PAHs}$ [3.]	**25** (123)	**33** (76)	**35** (20)	**49** (33)	**100** (14)	266
PEC-Q$_{totalPCBs}$ [4.]	**20** (98)	**25** (61)	**47** (43)	**47** (34)	**73** (30)	266
Hyalella azteca 28- to 42-day tests					**> 1.0**	
Mean Overall PEC-Q	**4** (45)	**6** (18)	**50** (18)	NC [5.]	**100** (28)	109
PEC-Q$_{metals}$	**5** (40)	**25** (24)	**60** (33)	NC	**100** (12)	109
PEC-Q$_{total\ PAHs}$	**8** (57)	**64** (37)	**55** (9)	NC	**100** (6)	109
PEC-Q$_{total\ PCBs}$	**4** (26)	**6** (35)	**17** (12)	NC	**97** (36)	109
Chironomus spp. 10- to 14-day tests					**> 5.0**	
Mean Overall PEC-Q	**29** (21)	**35** (78)	**35** (26)	**50** (34)	**78** (18)	177
PEC-Q$_{metals}$	**8** (12)	**43** (107)	**22** (36)	**75** (12)	**90** (10)	177
PEC-Q$_{total\ PAHs}$	**26** (64)	**33** (73)	**77** (13)	**85** (20)	**71** (7)	177
PEC-Q$_{total\ PCBs}$	**48** (58)	**23** (31)	**34** (32)	**35** (34)	**68** (22)	177

1. Mean Overall PEC-Q = Based on samples where average metal quotient, total PAH quotient, and PCB quotient summed and divided by 3.

In samples where the metals, total PAHs, and total PCBs were all measured, each of the three PEC-Qs were evaluated individually to determine their predictive ability, yielding the individual PEC-Q values below.

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2. PEC-Q$_{metals}$ = Average PEC quotient for the number of metals involved calculated .
3. PEC-Q$_{total\ PAHs}$ = Based on the samples where individual PAHs measured in samples which were summed to yield a total PAHs value.
4. PEC-Q$_{total\ PCBs}$ = Based on samples where total PCBs measured in samples.

5. NC = Not calculated.

Appendix A (continued)

Observations from Ingersoll et al. (2001):

- There was an overall increase in the incidence of toxicity with an increase in the mean quotients in toxicity tests involving all three test organisms.
- A consistent increase in the toxicity in all three tests occurred at a mean quotient of > 0.5. However, the overall incidence of toxicity was greater in the *Hyalella azteca* 28-day test compared to shorter term tests. The longer term tests, in which survival and growth are measured, tend to be more sensitive than the shorter term tests, with the acute to chronic ratios on the order of six indicated for *Hyalella azteca*.
- The use of chronic laboratory toxicity tests better identified chemical contamination in sediments compared to many of the commonly used measures of benthic invertebrate community structure. The use of longer-term toxicity tests in combination with SQGs may provide a more sensitive and protective measure of potential toxic effects of sediment contamination on benthic communities compared to use of the 10-day toxicity tests.
- There appears to be different patterns of toxicity when the PEC-Qs for the chemical classes are used alone or combined. The different patterns in toxicity may be the result of unique chemical signals associated with individual contaminants in samples. While the combined mean PEC quotient value from the chemical classes can be used to classify samples as toxic or nontoxic, individual PEC quotients of each chemical class might be useful in helping identify substances that may be causing or substantially contributing to the observed toxicity.
- The results of the evaluation indicate that the consensus-based PECs can be used to reliably predict toxicity of sediments on both a regional and national basis.

Example Calculation

The analytical results for a sediment sample and the steps to derive a mean overall PEC-Q for all the contaminants are as follows:

mg/kg dry wt.									
Sample Bulk Sediment Concentrations									
Metals							**Organics**		
Arsenic	Cadmium	Copper	Chromium	Lead	Nickel	Zinc	Total PAHs	Total PCBs	TOC
75	9	170	90	270	65	320	108	9.2	2.5%
Since TOC does not play a major role in the partitioning of metals from the sediments to the sediment pore water and its subsequent bioavailability, it is not necessary to convert metals concentrations to a dry weight normalized concentration at 1% TOC. Use the bulk sediment concentration as reported on the lab sheets to compare directly with the PEC SQGs. Normalization of metals concentrations to the fine fraction is done for the purposes of comparing the study site metal concentrations with the reference site concentrations on a common basis and is not related to the SQGs.							Convert the PAH and PCB concentrations dry wt. normalized concentrations at 1% TOC. Divide concentrations by 2.5. Step 2 above.		
75	9	170	90	270	65	320	43.2	3.68	
Determine the PEC concentrations for each contaminant (from Tables 1, 2, and 3 above).									
33	5	150	110	130	49	460	22.8	0.68	
Calculate the PEC-Q for each contaminant. Step 2 above.									
2.27	1.8	1.13	0.82	2.08	1.33	0.70	1.89	5.41	
Calculate a mean PEC-Q for the metals. Step 3 above.									
1.45							1.89	5.41	
Calculate an overall mean PEC-Q value from the 3 chemical classes (metals, PAHs, and PCBs). Step 4 above.									
Mean PEC-Q = 2.92									
Compare the 2.92 value with the ranges of PEC-Q values in the table above. For the shorter-term toxicity tests with *Hyalella azteca* and *Chironomus spp*., a value of 2.92 is in a range where 50% of the samples were toxic. For the longer-term tests with *H. azteca*, all of the samples were toxic at the PEC-Q value of 2.92. It appears based on these results, *H. azteca* or benthic organisms of similar sensitivity in the field populations may be significantly impacted by the concentrations of contaminants present. If these results represented an actual site, further assessments of the site is warranted.									

Observations From MacDonald et al. (2000)

MacDonald *et al.* (2000) also looked at the predictive ability of the CBSQGs. To examine the relationships between the degree of chemical contamination and probability of observing toxicity in freshwater sediments, the incidence of toxicity within various ranges of mean PEC quotients was calculated from an existing database. The data were plotted in a graph (Table 1, MacDonald *et al*. 2000). The interpolated data from this graph is in the table below. MacDonald et al. found that subsequent curve-fitting indicated that the mean PEC-quotient is highly correlated with incidence of toxicity (r^2 = 0.98), with the relationship being an exponential function. The resulting equation ($Y = 101.48\ (1-0.36^X)$ can be used to estimate the probability of observing sediment toxicity at any mean PEC quotient.

Relationship between Mean PEC Quotient and Incidence of Toxicity in Freshwater Sediments (Derived and Interpolated from MacDonald *et al.* 2000a)

Mean PEC Quotient	Average Incidence of Toxicity (%)
0	0
0.25	20
0.50	40
0.75	54
1.00	64
1.25	70
1.50	77
1.75	84
2.00	87
2.25	90
2.50	92
2.75	95
3.00	96
3.25	98
3.50	99
3.75	99.5
≥ 4.00	100

Utilizing the mean PEC-Quotient of 2.92 calculated in the example above yields a predicted average incidence of toxicity of approximately 95% based on the table immediately above. The chances are likely that if a sampled site yields a mean PEC-Q of 2.92, significant toxicity to infaunal species will be present.

Appendix B

Recommended Procedure for Calculating the Maximum Probable Background Concentration (MPBC) For a Metal or Organic Compound at Reference or Background Sites

Calculating the 95% upper confidence limit (UCL) of the mean of a data set of background concentrations for a parameter. Use of the UCL as the maximum probable background concentration (MPBC) for comparison purposes with the study site concentrations (Adapted from EPA, 1992b).

> Statistical confidence limits are a tool for addressing uncertainties of a distribution average. The 95% UCL of the arithmetic mean concentration is used as the average concentration because it is not possible to know the true mean. The 95% UCL therefore accounts for uncertainties due to limited sampling data. As sample numbers increase, uncertainties decrease as the UCL moves closer to the true mean. Sampling data sets with fewer than 10 samples may provide a poor estimate of the mean concentration (i.e., there is a large difference between the sample mean and the 95% UCL). Data sets with 10 to 20 samples may provide a somewhat better estimate of the mean (i.e., the 95% UCL is close to the sample mean). In general, the UCL approaches the true mean as more samples are included in the calculation.

Transformation of the Data

> The data set for the background concentrations should be looked at to determine if the data is lognormally or normally distributed. A statistical test should be used to identify the best distributional assumption for the data set. The W-test (Gilbert, 1987) is one statistical method that can be used to determine if a data set is consistent with a normal or lognormal distribution. In all cases, it is useful to plot the data to better understand the parameter distribution in the background or reference site area.

Assuming the data set for the background concentrations is normally distributed, the 95% UCL is calculated by the following four steps:

1) Calculate the arithmetic mean of the untransformed data.
2) Calculate the standard deviation of the untransformed data.
3) Determine the one-tailed *t*-statistic (see a statistical text for the Student *t Distribution* table).
4) Calculate the UCL using the following equation:

UCL = x + *t (s / square root of n)*

Where;
UCL = Upper Confidence Level of the Mean to be used as the maximum probable background concentration (MPBC).
x = Mean of the data
s = Standard deviation of the data
t = Student-t statistic from statistical textbook
n = number of samples

APPENDIX B (continued)

Example Calculation

10 samples were taken at a background site for mercury that had comparable hydrologic and sediment characteristics as the site under study but was not influenced by the sources of mercury contamination at the study site. The background sample concentrations for mercury were: 15, 30, 33, 55, 62, 83, 97, 104, 125, and 155 ug/kg.

Following the 4 steps above –

1) Mean mercury concentration - 75.9 ug/kg
2) Standard deviation – 45.02
3) Student t-statistic value for one-tail test. n = 10 samples. Degrees of freedom 10 – 1 = 9.
   t-distribution - 1.833
4) UCL = x + *t (s / square root of n)*
   UCL = 75.9 + 1.833 (45.02 / square root of 10)
   UCL = 75.9 + 1.833 (45.02 / 3.16)
   UCL = 75.9 + 1.833 (14.25)
   UCL = 75.9 + 26.12
   UCL = 102.02 ug/kg

   The UCL value for mercury of 102.02 ug/kg becomes the maximum probable background concentration (MPBC) that will be used to compare the study site concentrations against. Concentrations of mercury in study site sediment samples that are greater than the 102.02 ug/kg value can be considered to be influenced by the sources of mercury other natural or ubiquitous (e.g., atmospheric depositions) sources. As discussed above in the main body of this document, the percent fine fractions need to be looked at in the sediment samples under comparison. If the relative contribution of fines are the same in the samples from the background site and the study site, then no adjustments need to be made. If the percent fines are significantly different between the samples and the sites, then considerations for normalization of the mercury concentrations to the fine content should be looked at in order to do relevant site-to-site comparisons of metal concentrations.

   The CBSQG TEC value for mercury is 180 ug/kg (Table 1 above). The MPBC for mercury in this example at 102.02 ug/kg is less than the MPBC value. An interpretation of this relationship is that benthic macroinvertebrates are possibly tolerant of mercury concentrations that are somewhat greater than background concentrations. This relationship may come into play if a decision is made to use the greater of the MPBC or the TEC value to drive the cleanup of a site.

An example of what fewer background samples would mean to the resulting MPBC value can be seen by the following example using only 4 of the sample results for mercury – 30. 62, 104, and 155 ug/kg.

1) Mean mercury concentration – 87.8ug/kg
2) Standard deviation – 54.11
3) Student t-statistic value for one-tail test for n = 4 samples. Degrees of freedom 4 – 1 = 3
   t-distribution – 2.353
   UCL = x + *t (s / square root of n)*
   UCL = 87.8 + 2.353 (54.11 / square root of 4)
   UCL = 87.8 + 2.353 (54.11 / 2)
   UCL = 87.8 + 2.353 (27.06)
   UCL = 87.8 + 63.7
   UCL = 151.5 ug/kg

APPENDIX C
Notes on Dioxins and Furans

- Polychlorinated dibenzo-p-dioxins and dibenzofurans are ubiquitous contaminants, primarily from combustion sources. Background concentrations are normally in the range 0.15 - 2.5 pg TCDD-EQ/g Sediment.
- There are concerns with the other 2,3,7,8-substituted congeners beside 2,3,7,8-TCDD and TCDF. There is a need to request that all 17 - 2,3,7,8 substituted congeners be analyzed for. Analytical costs are high. To do an adequate environmental assessment, detection levels for 2,3,7,8-TCDD need to be at the single digit pg/g level.
- Dioxins and furans are not produced commercially but are unintended by-products from various chemical manufacturing and other sources.
- Dioxins and furans are found in discharges from wood treatment facilities that use pentachlorophenol, kraft pulp mills, and chemical manufacturing plants that produced pentachlorophenol, trichlorophenol, and the pesticides 2,4-D and 2,4,5-T. Also, if a water body has a history of aquatic applications of the herbicide Silvex, residual dioxins and furans may be present
- For some perspective, the department's landspreading program for paper mill sludges sets limits for spreading based on land uses - Silviculture - 10 pg/g; Agriculture - 1.2 pg/g; Grazing - 0.5 pg/g.
- Examples of high levels of dioxins/furans at Wisconsin sediment sites include - Crawford Creek - discharge from wood treatment facility that used pentachlorophenol - 5,500 pg TCDD-EQ/g; Military Creek-discharge from wood treatment facility that used pentachlorophenol– 2,500 pgTCDD-EQ/g; Fox River - paper mill discharges - 21 - 441 pg TCDD-EQ / g; and Wisconsin River **-** paper mill discharges - 31 - 78 pg TCDD-EQ / g.
- The recommendation is that dioxin and furan analysis only be done where there is a demonstrated need given the identification of possible historical sources at a site.
- The different 2,3,7,8 – substituted dioxins and furans have toxic equivalency factors (TEF) assigned to them relative to their toxicity compared to 2,3,7,8-TCDD. The table below provides a method to calculate the summed TCDD equivalent concentration for all the substituted forms in a sample.

2,3,7,8 - Substituted Dioxin and Furan Congeners

Worksheet For Calculating 2,3,7,8-TCDD Equivalent Concentrations	Sediment Concentration pg/g (ppt) dry weight	Toxic Equivalency Factors (TEF) (Equivalency to 2,3,7,8-TCDD)	pg/g x TEF = Toxic Equivalency to 2,3,7,8-TCDD Or TCDD-EQ
Dioxins			
2,3,7,8-TetraCDD		1.0	
1,2,3,7,8-PentaCDD		0.5	
1,2,3,4,7,8-HexaCDD		0.1	
1,2,3,6,7,8-HexaCDD		0.1	
1,2,3,7,8,9-HexaCDD		0.1	
1,2,3,4,6,7,8-HeptaCDD		0.01	
OctaCDD		0.001	
Furans			
2,3,7,8-TetraCDF		0.1	
2,3,4,7,8-PentaCDF		0.5	
1,2,3,7,8-PentaCDF		0.05	
1,2,3,4,7,8-HexaCDF		0.1	
1,2,3,6,7,8-HexaCDF		0.1	
2,3,4,6,7,8-HexaCDF		0.1	
1,2,3,7,8,9-HexaCDF		0.1	
1,2,3,4,6,7,8-HeptaCDF		0.01	
1,2,3,4,7,8,9-HeptaCDF		0.01	
OctaCDF		0.001	
Sum of TCDD-EQ of Individual Substituted Dioxin and Furan Congeners (___pg TCDD-EQ / kg sediment) =			

APPENDIX D

Dry Weight Sediment Concentrations of Organic Compounds Normalized to 1% TOC for Comparison with CBSQGs and Grain Size Normalizations of Metals for Site-to-Site Comparisons						
Sample Site:		**Example Calculations** (Request a copy of Excel Spreadsheet)				
Sample Description:						
Date:						
ug/g = ppm = mg/kg						
ng/g = ppb = ug/kg						
TOC reported as mg/kg ÷ 10,000 = % TOC						
	Bulk Chemistry					
Parameter	**Concen-tration**	**Units**	**% TOC in Sample**			
TOC	25,000	mg/kg	2.5%			
Dry Wt. Concentration ÷ TOC expressed as a % = Concentration Normalized to 1% TOC						
PAHs	**Dry Weight Concentration**		**Normalized to 1% TOC for Comparison With CBSQG Values**			
Acenapthene	3.2	ug/kg	1.3	ug/kg @ 1% TOC		
Acenaphthylene	5.9	ug/kg	2.4	ug/kg @ 1% TOC		
Anthracene	57.2	ug/kg	22.9	ug/kg @ 1% TOC		
Fluorene	77.4	ug/kg	30.9	ug/kg @ 1% TOC		
Napthalene	176	ug/kg	70.4	ug/kg @ 1% TOC		
2-Methylnapthalene	20.2	ug/kg	8.1	ug/kg @ 1% TOC		
Phenanthrene	204	ug/kg	81.6	ug/kg @ 1% TOC		
Benzo(a)anthracene	108	ug/kg	43.2	ug/kg @ 1% TOC		
Benzo(a)pyrene	150	ug/kg	60	ug/kg @ 1% TOC		
Benzo(e)pyrene	150	ug/kg	60	ug/kg @ 1% TOC		
Benzo(b)fluoranthene	240	ug/kg	96	ug/kg @ 1% TOC		
Benzo(k)fluoranthene	240	ug/kg	96	ug/kg @ 1% TOC		
Benzo(g,h,i)perylene	170	ug/kg	68	ug/kg @ 1% TOC		
Chrysene	166	ug/kg	66.4	ug/kg @ 1% TOC		
Dibenz(a,h)anthracene	33	ug/kg	13.2	ug/kg @ 1% TOC		
Fluoranthene	423	ug/kg	169.2	ug/kg @ 1% TOC		
Indeno(1,2,3-c,d)pyrene	200	ug/kg	80	ug/kg @ 1% TOC		
Pyrene	195	ug/kg	78	ug/kg @ 1% TOC		
Total PAHs (sum of 18 PAHs listed above)	2618.9	ug/kg	1,047.6	ug/kg @ 1% TOC		

PCB and Pesticides	Concen-tration	Units	Normalized to 1% TOC for Comparison With CBSQG Values	
PCBs (total)	60	ug/kg	21	ug/kg @ 1% TOC
Aldrin	2	ug/kg	0.8	ug/kg @ 1% TOC
BHC	3	ug/kg	1.2	ug/kg @ 1% TOC
a-BHC	6	ug/kg	2.4	ug/kg @ 1% TOC
B-BHC	5	ug/kg	2	ug/kg @ 1% TOC
Y-BHC (lindane)	3	ug/kg	1.2	ug/kg @ 1% TOC
Chlordane	3.2	ug/kg	1.3	ug/kg @ 1% TOC
Dieldrin	1.9	ug/kg	0.8	ug/kg @ 1% TOC
Sum pp DDD	4.9	ug/kg	1.9	ug/kg @ 1% TOC
Sum pp DDE	3.2	ug/kg	1.3	ug/kg @ 1% TOC
Sum op + pp DDT	4.2	ug/kg	1.7	ug/kg @ 1% TOC
Sum of DDT and metabolites	5.3	ug/kg	2.1	ug/kg @ 1% TOC
Endrin	3	ug/kg	1.2	ug/kg @ 1% TOC
Heptachlor Epoxide	2.5	ug/kg	1.0	ug/kg @ 1% TOC
Mirex	7	ug/kg	2.8	ug/kg @ 1% TOC
Toxaphene	1	ug/kg	0.4	ug/kg @ 1% TOC

Metals

Particle Size				
	% sand	50	%	
	% silt	25	%	Fine Fraction Silt + Clay = 50% or 0.50
	% clay	25	%	

Dry Wt. Concentration ÷ Fines expressed as decimal fraction = Normalized to Fine Concentration

Metals	Dry Weight Concentration (Compare with CBSQGs		Normalized to Fine Concentration for Site-to-site Comparisons(Not for Comparison with CBSQGs)	
Antimony	2	mg/kg	4	mg/kg fines
Arsenic	9.8	mg/kg	19.6	mg/kg fines
Cadmium	0.99	mg/kg	1.98	mg/kg fines
Chromium	43	mg/kg	86	mg/kg fines
Copper	32	mg/kg	64	mg/kg fines
Iron	20,000	mg/kg	40,000	mg/kg fines
Lead	36	mg/kg	72	mg/kg fines
Manganese	460	mg/kg	920	mg/kg fines
Mercury	0.18	mg/kg	0.36	mg/kg fines
Nickel	23	mg/kg	46	mg/kg fines
Silver	1.6	mg/kg	3.2	mg/kg fines
Zinc	120	mg/kg	240	mg/kg fines

Appendix E

Identification of Contamination that Leads to Adverse Effects

Contamination of a chemical nature (i.e., a contaminant) is a substance or substances (either organic or inorganic) that are present in environmental media such as sediments or surface waters that are found above levels that would normally occur. What is normal or background for metals or nutrients (e.g., nitrogen, phosphorus) would be those metals and nutrients at levels that originate from the natural soil types and the geochemical components of the watershed. What is normal for natural organic compounds would generally be those compounds that originate from natural watershed-source vegetative or animal matter that are deposited on the bottoms of lakes, streams, and wetlands. Organic chemicals manufactured by humans and released to the environment by various mechanisms generally do not have counterparts found in nature and therefore any levels found in environmental media would be considered potential contamination. Many manufactured organic compounds may be found ubiquitously at low levels in sediments especially in urban areas.

Environmental concerns arise when the level of contamination (concentration of contaminants) in surface waters and sediments leads to observed and measurable effects to biological receptors, such as 1) chronic and/or acute toxicity (the contaminant becomes a toxicant) to aquatic receptors (for example directly to aquatic life such as bottom inhabiting macroinvertebrates), and/or 2) concerns about humans and wildlife that are upper food chain organisms who may become exposed to harmful levels of contaminants principally through consumption of aquatic organisms that have bioaccumulated the contaminants. For the toxicity to aquatic organisms to be realized and/or unacceptable levels of bioaccumulation to occur, the aquatic organism has to (a) be exposed to the potential toxicant in its habitat, (b) the potential toxicant has to be in a form available for uptake, and (c) the uptake or dose of the contaminant has to be at a level that causes toxicity to the particular exposed receptor or results in levels of bioaccumulation that may pose risks to humans and/or wildlife who consume the exposed receptor as food.

Elevated levels of nutrients can lead to eutrophication of water bodies and production and deposition plant materials in sediments that deplete oxygen levels in the water body when they decompose. Addition and decomposition of natural organic matter and anthropogenic-added organic matter in sediments can lead to production of hydrogen sulfide and ammonia levels that may be detrimental to benthic organisms.