Human Health-based Water Quality Standards Technical Support Document

Water Quality Standard Amendments – Minn. R. chs. 7050 and 7052 [Final]

Minnesota Pollution Control Agency

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Revision History
An earlier version of this technical support document served as an exhibit in adopting revised human health methods into Minn. R. chs. 7050 and 7052. As a result of comments received and the need to make minor rule language changes during the rulemaking (39SR3144), the MPCA has revised the 2014 technical support document to reflect the final discussion of the technical foundation for Class 2 human health-based water quality standards. If there are discrepancies between rule language cited in this technical support document and the rules as adopted, the MPCA intends that the adopted rule language is what is justified. The use of the word "proposed" was relevant at the time the document was first published as a draft. The methods in this document reflect the final adopted rule.

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Acronyms and Abbreviations

ADAF	Age Dependent Adjustment Factor
$AF_{Lifetime}$	Adjustment Factor-Lifetime
ATSDR	Agency for Toxic Substance and Disease Registry
AWQC	Ambient Water Quality Criteria (EPA)
BAF	Bioaccumulation Factor
BCC	Bioaccumulative Chemical of Concern
BCF	Bioconcentration Factor
BMD	Benchmark Dose
BSAF	Biota-Sediment Accumulation Factor
BW	Body Weight
C	Carcinogen, Linear
CR	Cancer Risk level
CC_{dev}/CS_{dev}	Chronic Criterion or Standard – New less-than-chronic evaluation for drinking water use classes
CC_{df} /CS_{df}	Chronic Criterion or Standard – 1990 for drinking water, fish consumption, and recreation use classes
CC_{dfr} /CS_{dfr}	Chronic Criterion or Standard – Revised for drinking water, fish consumption, and recreation use classes
CC_f/CS_f	Chronic Criterion or Standard- 1990 for fish consumption use class
CC_{ft}/CS_{ft}	Chronic Criterion or Standard – New for fish tissue-based
CC_{fr}/CS_{fr}	Chronic Criterion or Standard – Revised for fish consumption and recreation use class
CSF	Cancer Slope Factor (same as q1*)
CSFII	Continuing Survey of Food Intakes by Individuals (U.S. Dept. of Agriculture)
CWA	Clean Water Act
DAF	Dosimetric Adjustment Factors
DC	Domestic Consumption (Class 1 Water Quality Standard)
DWIR	Drinking Water Intake Rate
EPA	U.S. Environmental Protection Agency
FAV	Final Acute Value (Class 2 Water Quality Standard)
FCA	Fish Consumption Advice (MDH)
FCR	Fish Consumption Rate
FDA	U.S. Food and Drug Administration
GLI	Great lakes Initiative (Minn. R. ch. 7052)
HA	Health Advisory (SDWA)
HBV	Health Based Value; developed by the Minnesota Department of Health (MDH) using the same methodologies as HRLs
HED	Human Equivalent Dose
HH-WQS	Human Health-based Water Quality Standard
HRL	Health Risk Limits; drinking water standards from MDH
IR	Intake Rate (drinking water-MDH)

IRIS	Integrated Risk Information System
IWR	Incidental Water Intake Rate
MCL	Maximum Contaminant Level (SDWA)
MOA	Mode (or mechanism) of Action
MS	Maximum Standard (Class 2 Water Quality Standard)
MDH	Minnesota Department of Health
MPCA	Minnesota Pollution Control Agency
NHANES	National Health and Nutrition Examination Survey
NLC	Nonlinear Carcinogen
NPDES	National Pollutant Discharge Elimination System
POD	Point of Departure
q1*	Carcinogen potency factor (slope) (1990)
RAGS	Risk Assessment Guidance for Superfund
RfD	Reference Dose for noncancer toxicants and nonlinear carcinogens
RSC	Relative Source Contribution factor
SDWA	Safe Drinking Water Act
SONAR	Statement of Need and Reasonableness
TL	Trophic Level (with BAF)
UF	Uncertainty Factor
USEPA	U.S. Environmental Protection Agency
WOE	Weight of Evidence
WPC	Water Pollution Control (first Minnesota water quality rules series)
WQS	Water Quality Standard

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Introduction

Minnesota's human health-based water quality standards (HH-WQSs) provide protections for the beneficial uses of drinking water, fish consumption, and recreation. HH-WQSs were first adopted in 1990; since that time there have been significant improvements in risk assessment methods and policies both at the State and Federal levels. The revisions being considered to the HH-WQS methods in Minnesota's water quality rules (Minn. R. chs. 7050 and 7052) center on recent U.S. Environmental Protection Agency (EPA) guidance and Minnesota Department of Health's (MDH) Health Risk Limits (HRL) rule. These revisions incorporate the latest public health practices for development of HH-WQSs that when fully implemented will enhance protection for human users of Minnesota's surface waters. Like all WQSs, they serve as the basis for setting wastewater discharge effluent limits, assessing water quality, and determining if waters are impaired (do not meet WQS).

New toxicological and exposure parameters

HH-WQSs represent numeric standards adopted into rule as Class 2 Chronic Standards (CS) or applied as site-specific Chronic Criteria (CC) for pollutants detected in surface water that lack listed CSs (Minn. R. 7050.0217 to 7050.0218 and Minn. R. 7052.0110). The general algorithms (equations and steps) for developing HH-WQS[1] encompass the toxicological profile of a pollutant and representative exposure rates for people using surface waters as a source of drinking water and for recreating and fishing. This information is incorporated into algorithms to develop pollutant-specific numeric standards or criteria that minimize the risk of adverse effects from long-term (chronic) exposure. The fundamental formula for developing Class 2 HH-WQS is:

Class 2 HH-WQS, Chronic Standard or Criterion (CS/CC) =

Toxicological Value / Exposure

where, the *Toxicological Value* is either based on protection from noncancer (or nonlinear carcinogenic) adverse effects using a reference dose (RfD) or linear carcinogenic potency expressed by a slope factor (q1* or CSF), with new adjustment factors for higher early-life risk. *Exposure* encompasses the intake rates and other parameters used to estimate and limit how much of a pollutant a person may ingest relevant to the three beneficial uses.

Revisions to toxicological parameters: MPCA continues to strive for consistency with MDH's HRL and risk assessment methods. Toxicological values used in HH-WQSs are expressed as RfDs

[1] To assist in describing the Class 2 CS or CC based on human health protection the alternate term "Human Health-based Water Quality Standard" (HH-WQS) is also used. Water quality standards also include a Class 1 designation for drinking water protection for all groundwater and for Class 2A and 2Bd surface waters; the Class 1, Domestic Consumption (DC) standards are the Federal Safe Drinking Water Act standards incorporated by reference. They are based on different methods and not currently developed by MPCA. A full discussion is found in Section II. A.

for noncancer effects and cancer potency slope factors (CSFs) and will be from MDH's HRL rule or risk based guidance or developed following MDH toxicological methods as incorporated in the revised HH-WQS methods. Therefore, if EPA publishes toxicological values, those values would only be used following evaluation and completion of any needed modifications based on the revised HH-WQS methods.

- MDH now develops RfDs for four durations (acute, short-term, subchronic, and chronic); MPCA's HH-WQSs are Chronic Standards (CS) developed based on protection level goals with "lifetime" considerations. MPCA is adding the use of less-than-chronic (acute to subchronic) duration RfDs on a pollutant-specific basis when appropriate to ensure the CS is protective for shorter duration health effects; these shorter durations may have more stringent final standards, especially when the RfDs are matched with MDH's higher, shorter duration drinking water intake rates (DWIRs) as outlined in *Revisions to Exposure Parameters.*
- Improved toxicological evaluations by MDH and EPA also mean that characterization of a pollutant's adverse effects will better account for its mode of action, target organs or systems, and nonlinear or linear carcinogenic characteristics. As an example, MDH lists *Health Endpoints* in the HRL rule by target organ or system. These endpoints will be added to HH-WQSs to account for mixtures of pollutants that act on common *Health Endpoints.*
- Reviews by MDH and EPA clearly demonstrate that exposure to linear carcinogens in early life leads to higher cancer risk as measured by a number of variables (e.g., higher incidence, earlier neoplasm formation, etc.). New CSF adjustments will be used to address this higher risk on either a pollutant-specific basis (Lifetime Adjustment Factor) or by a default approach using Age Dependent Adjustment Factors (ADAFs).

Revisions to exposure parameters: MPCA plans to propose new exposure intake rates for drinking water, incidental water, and fish consumption:

- Revised intake rates will directly incorporate body weight; including body weight improves the accuracy of the rates, reflects newer exposure data, and makes comparison between rates easier.
- DWIRs are from MDH's HRL rule based on higher percentile (95^{th}), time-weighted averages, based on newer data, and considerations of life stage differences.
- MPCA has revised the incidental water rate based on children, reasonable maximum swimming activity patterns (used at Minnesota site assessments), and minimum defined chronic duration (>10% of a lifetime or 8 years).
- Clarification of the existing fish consumption rate (FCR) of 30 g/d with a 70 kg body weight as representative of adults and applied as the chronic FCR of 0.43 g/kg-d. MPCA recently reviewed available data and policy to develop a children's FCR of 0.86 g/kg-d for ages 1 through 5 for use with the age groups defined when applying the new ADAFs for linear carcinogens.

***Additional exposure-related parameters**:* Relative Source Contribution (RSC) factors provide a means to account for other exposure routes and sources of the pollutant for the population addressed by HH-WQSs and to limit that exposure below the RfD. EPA's RSC Exposure Decision Tree (2000) adds more clarification and guidance on how to develop RSCs. EPA's approach also includes a new default value of 0.5 (50%) to add to the previously used single default option of 0.2 (20%). MDH also uses this approach for HRLs.

The 2000 EPA guidance also includes methods to develop bioaccumulation factors (BAFs). Minn. R. ch. 7052 already has comparable BAF methods, because those methods, developed as part of the Great Lakes Initiative (GLI), served as the foundation for the new national BAFs. MPCA is planning to update the BAF methods in Minn. R. ch. 7050 based on EPA's guidance, consistency with Minn. R. ch. 7052, and use of available Minnesota or regional data.

- MPCA is planning to revise methods in Minn. R. ch. 7050 to address Trophic Levels (TL) distinctions in fish species consumed and their average lipid content as done in Minn. R. ch. 7052 (76% TL4, 24% TL3). This approach better refines BAFs by addressing species differences that can affect the concentration of a pollutant in fish tissue.
- Another set of defaults apply in the BAF methods based on organic carbon fractions and concentrations in surface waters; dissolved and particulate organic carbon (DOC and POC) affect the uptake of pollutants in aquatic organisms. Plans are to use average values from Minnesota lakes and streams for statewide use and retain the GLI values in the Lake Superior basin.

New approach for environmental degradates: MDH's HRL methods include language to address a pollutant's breakdown products or environmental degradates (Minn. R. 4717.7900); when toxicological data are lacking to develop a HRL or health-based guidance value (i.e., HBV) for degradates the "parent" pollutant's HRL or HBV is used to characterize risk. MPCA plans to adopt the same approach for HH-WQSs.

Addition of fish tissue algorithms

Until 2008, all WQSs were developed and applied as surface water concentrations. As part of the recent revisions to Minn. R. ch. 7050 MPCA adopted a fish tissue standard for mercury based on EPA's 2001 national criterion. Key reasons for adding a HH-WQS in this medium:

- Eliminates the need to estimate BAFs, which are highly variable for mercury.
- Fish consumption (marine and freshwater) is the primary source of mercury exposure to humans.
- Most data on mercury in surface waters come from fish tissue monitoring.

To address similar issues for other pollutants, MPCA is proposing to adopt methods to develop additional fish tissue standards for pollutants that bioaccumulate in fish muscle tissue:

- Ultimately fish consumption is the route of highest exposure for highly bioaccumulative pollutants, such as polychlorinated biphenyls (PCBs) and legacy pesticides (i.e., dieldrin).
- The algorithms are already established in EPA's fish consumption advisory program and 2001 mercury criterion (see below).

- Almost all data available on bioaccumulative pollutants (generally pollutants with BAFs > 1,000) are from fish tissue sampling, not water monitoring.
- These numeric HH-WQSs will supplement the existing narrative standard based on MDH Fish Consumption Advice (Minn. R. ch. 7050.0150, subp. 7).

Fish Tissue-based algorithms in mg/kg or ppm (part-per-million) = RfD x RSC / $FCR_{chronic}$

Or in µg/kg or ppb (part per billion) = RfD x RSC / $FCR_{chronic}$ x 1000 µg/mg

Results

The draft parameters outlined above will be incorporated into algorithms specific to surface water use classifications (Class 2A, 2Bd, and 2B/2C/2D) and toxicological profiles (noncarcinogen, nonlinear carcinogen, and linear carcinogen). These revised procedures for developing HH-WQSs are proposed to replace those listed in Minn. R. chs. 7050 and 7052 for developing CSs (or CCs) applicable to water concentrations and add supplemental algorithms for fish tissue-based CSs (and CCs). HH-WQSs are based on pollutant-specific data, so as a whole there isn't one statement that can be made as far as if these revised methods will result in less or more stringent CSs. While some parameters will tend to decrease the CS (e.g., higher DWIRs and application of cancer potency adjustment factors), if newer toxicological values are less stringent than were used in the current rule the final CS could be less stringent. MPCA plans to update pollutant-specific HH-WQSs (CSs) and site-specific CCs based on final, recently updated toxicological reviews promulgated for HRLs in Minn. R. 4717.7500 through 4717.7900, published as Health-based Guidance (HBVs), or developed by MPCA following MDH methods as incorporated into the revised HH-WQS methods.

Application of Revised Methods

WQSs are pollutant specific; therefore, standards are needed for a large spectrum of toxic substances that potentially impact people, aquatic life, or fish-eating wildlife. Historically, the toxic pollutants with WQS have centered on the Clean Water Act (CWA) identified *Priority Pollutants,* and in the Great Lakes, *Bioaccumulative Chemicals of Concern* (BCCs). These types of pollutants comprise the majority of 69 toxic chemicals with CSs listed in Minn. R ch. 7050 and 29 in Minnesota R. ch. 7052. For about half of these standards, human health protection is the most stringent numeric CS as identified in Minn. R. 7050.0222 and Minn. R. 7052.0100. The pollutants listed in rule do not cover all known or potential toxic pollutants that could be present in surface water and fish; therefore, methods are adopted in both rules to develop a site-specific CC for any toxic pollutant that has sufficient data. The CC serves the same purpose as CS with specific implementation requirements described in Minn. R. 7050.0218 and 7052.0110.

Future application of revised methods for pollutant-specific CSs and site-specific CC:

MPCA will implement the proposed revised methods by replacing existing pollutant-specific numeric CS with updated standards in in the next phase of HH-WQS rule revisions. However, if MPCA determines through consultation with MDH that updated CSs are needed sooner for application on a site-specific basis, MPCA can modify a standard with EPA approval as described in Minn. R. 7050.0220, subp. 7 and 7052.0270. Wider application would mean also considering this change in subsequent Triennial Standard Reviews. All future site-specific CC will be developed based on the revised methods.

Uncertainty Analysis

HH-WQS are evaluated and updated to meet the State's protection level goals and CWA requirements for ensuring waters meet the designated uses of drinking water, fish consumption, and recreation. A full discussion on how the human health methods are developed to minimize and account for uncertainty is found in Sections VI and VII. C.

Implementation of Human Health-based Water Quality Standards

The HH-WQS in Minn. R. chs. 7050 and 7052, along with other WQSs, provide the foundation for effluent limits in National Pollutant Discharge Elimination System (NPDES) wastewater and stormwater permits and assessment of available pollutant-specific monitoring data in surface waters and fish that may be of concern for human health. Implementation of new or revised effluent limits based on revised pollutant-specific HH-WQS will occur as described in application of revised methods, following the adoption of future CSs or development of new or updated site-specific CC. Implementation of the standards is broader than the scope of this document, but is founded in MPCA and other State agencies' programs and approaches to water resource protection. However, due to limitations in scientific understanding, available resources, and nonpoint pollutant sources, not all pollutants have control measures to achieve the WQS to meet beneficial uses in all waterbodies. MPCA assesses many pollutants with WQS and if they don't meet WQS list those waterbodies as impaired and develop studies (Total Maximum Daily Load or TMDL) to determine sources and reductions needed to meet WQS. The processes of identifying and listing impaired waters and completing TMDLs are required under the CWA. MPCA continually works with many State, federal, academic, and local partners, and the public to identify and eliminate impaired water and gaps in monitoring data and control measure to ensure HH-WQSs are being met. An overview of HH-WQS implementation and limitations is provided in Section I.D.

I. Introduction

A. Purpose and scope

The primary purpose of this document is to describe the basis for the current and revised methods used to develop Class 2 numeric water quality criteria and standards for human health protection from elemental and synthetic chemical contaminants characterized as toxic pollutants[2,3]. Water quality standards for the protection of other beneficial uses and populations for which Minnesota's surface waters are protected, such as microbiological water quality standards, and aquatic organism toxicity, are not covered in this document. Drinking water protection provided by Class 1 Domestic Consumption (DC) standards is discussed in this guidance, but is handled differently from Class 2 Human Health-based Water Quality Standards (HH-WQS).

The procedures described in this guidance relate to existing and revised methods for HH-WQS and proposed amendments to Minnesota Rules chapter (Minn. R. ch.) 7050, which are applicable statewide, and Minn. R. ch. 7052, which applies in the Lake Superior basin[4].

B. Overview of water quality standards and site-specific criteria

Beginning with Minnesota's first water quality rules in 1963 and enhanced by requirements of the 1972 Federal Water Pollution Control Act (Clean Water Act or CWA), and subsequent amendments to the CWA, the basis for protecting the quality of waters of the state[5] is in water quality standards (WQS). WQSs consist of three elements (USEPA 1994):

I. Classifying waters with designated beneficial uses;
II. Narrative and numeric standards to protect those uses; and
III. Nondegradation (antidegradation) policies to maintain and protect existing uses and high quality waters.

WQSs are thus the fundamental regulatory and policy foundation to preserve, assess, and restore the quality of all groundwater[6] and surface waters. The term "water quality standards" is commonly used in both a broad and narrow sense. Broadly speaking, WQSs include all the

[2] Minn. Statute 115: Toxic Pollutant: means those pollutants, or combinations of pollutants, including disease-causing agents, which after discharge and upon exposure, ingestion, inhalation or assimilation into any organism, either directly from the environment or indirectly by ingestion through food chains, will, on the basis of information available to the agency, cause death, disease, behavioral abnormalities, cancer, genetic mutations, physiological malfunctions, including malfunctions in reproduction, or physical deformation, in such organisms or their offspring.

[3] The terms *pollutant* and *chemical* are relied on in this document to describe a *toxic pollutant*. Other sources referenced in this document may refer to a *toxic pollutant* interchangeably as a *chemical, element, environmental agent, contaminant,* or *substance.*

[4] The technical support document (TSD) sets the foundation for proposing revised human health methods; additional method changes resulting from the rulemaking process will be included in an addendum.

[5] The term "waters of the state" is defined in Minn. Stat. § 115.01, subd. 22.

[6] In Minnesota, other statutes and rules also pertain to groundwater protection and quality.

three elements and legal requirements in water quality rules described above, including minimum wastewater treatment requirements and effluent limits for point source dischargers. In the more narrow sense, "standards" may refer only to the pollutant-specific numeric and narrative standards (element II) that define acceptable conditions for meeting the beneficial uses.

For toxic pollutants, the water quality rules also distinguish between "criteria" that are applied on a site-specific basis and the "standards" adopted through rulemaking for statewide application. When a pollutant that is lacking a promulgated numeric standard in Minn. R. 7050.0220, 7050.0222, 7050.0227, or Minn. R. 7052.0100 is detected in surface waters MPCA has promulgated methods to develop numeric "site-specific criteria." The "criteria" MPCA develops are based on the methods for a Final Acute Value (FAV), Maximum Criterion (MC), and Chronic Criteria (CC) described in Minn. R. 7050.0217 and 7050.0218[7] and Minn. R. 7052.0110.

The WQSs and site-specific criteria discussed in this document are for Class 2 designated surface waters, where State and CWA goals are integrated. As stated in 7050.0140, subp. 3:

> **Class 2 waters, aquatic life and recreation.** Aquatic life and recreation includes all waters of the state that support or may support fish, other aquatic life, bathing, boating, or other recreational purposes and for which quality control is or may be necessary to protect aquatic or terrestrial life or their habitats or the public health, safety, or welfare.

This guidance focuses on Class 2 Human Health-based Water Quality Standards (HH-WQSs) and site-specific CC. These methods serve as the basis for developing chronic or long-term protection for humans from toxic pollutants to ensure the beneficial uses of drinking water (where designated) and fish consumption and recreation in all surface waters are met. Drinking water protection in groundwater is addressed under the Class 1 designation and Domestic Consumption (DC) standards, which are the Federal Safe Drinking Water Act (SDWA) standards incorporated by reference. Application of Class 1 DC standards in surface waters designated for drinking water protection is discussed in Section II.A.

As background, numeric criteria and standards include three evaluations based on protections that examine different durations of potential pollutant exposure and most sensitive populations for the Class 2 beneficial use classifications. The aquatic life community is considered most at risk for acute toxicity; whereas chronic toxicity is relevant to aquatic life, human health, and fish-eating wildlife; WQSs are adopted into rule as the:

- Final Acute Value (FAV),
- Maximum Standard (MS), and
- Chronic Standard (CS).

CSs are calculated three possible ways:

- Based on direct toxicity to aquatic life (CS_{tox}),
- Based on human health impacts (CS_{dfr}, CS_{dev}, CS_{fr}, CS_{ft}) , and, less often,
- Based on impacts to fish-eating wildlife (CS_w).

[7] Adoption of the new HH-WQS methods will include a new part 7050.0219.

The FAV and MS are always aquatic life toxicity-based, never human health- or wildlife-based. Both are based on acute toxicity (mortality) data for fish and other aquatic organisms, and both are designed to protect aquatic organisms from short-term exposures (i.e., 24 hours or less) to pollutant concentrations above the CS. The MS always equals one-half the FAV[8].

CSs are designed to provide protection from harmful effects that might result from long-term or indefinite exposure to the pollutant, with the applicable duration for the CS being dependent on the population assessed. The CS based on toxicity to aquatic organisms applies on a shorter duration (a 4-day average) to reflect the very short lifespan of some aquatic species. The human health-based CS is applied as a 30-day average to reflect the much longer human lifespan (see Section VII for details on application). The revised methods being proposed in the accompanying human health methods will provide additional protection from health effects that can occur from shorter durations of exposure to a toxicant and continue to fit the 30-day average application duration. The lowest calculated value of the three CSs is listed in rule as the applicable WQS, although, all numeric CSs need to be met for their given durations.

In addition to toxicity and impacts on humans, when data are available the development of CSs based on human health can include environmental characteristics of toxic substances listed below:

- Bioaccumulation, including biomagnification of tissue concentrations of the chemical transferred to higher levels of the aquatic food chain,
- Taste and odor impairment of fish tissue (organoleptic WQSs), and
- Toxicity of environmental degradates formed from the original or parent pollutant by environment processes.

WQS have been developed as water concentrations that if not exceeded in magnitude, duration and frequency (once in three years for acute WQSs and twice in three years for CSs) protect beneficial uses, the exception being the supplementary mercury fish tissue-based HH-WQS adopted in 2008. In 2001 for mercury, EPA published the first criterion with a fish tissue concentration instead of water concentration. EPA had determined for many reasons that this was a better approach than using a water concentration for a highly bioaccumulative pollutant. MPCA adopted a fish tissue HH-WQS for mercury in 2008 and as described in this guidance is now including algorithms to develop fish tissue-based HH-WQSs for other bioaccumulative pollutants (see Sections IV.E. and V.B.). Fish-tissue based standards are applied based on magnitude only.

WQSs are pollutant specific; therefore, standards are needed for a large spectrum of toxic substances that potentially impact people, aquatic life, or fish-eating wildlife. The methods described in this guidance support the development of HH-WQSs, applied as CSs listed in Minn. R. chs. 7050 and 7052, and for future development of site-specific criteria. Site-specific criteria are numeric values developed according to methods listed in rule for use when a pollutant is

[8] Minn. R. ch. 7050.0222, subp. 7, item E, does have an approach to limit the magnitude of the MS and FAV; when their calculated values are greater than 100 times and 200 times the CS, respectively; those threshold concentrations are applied to ensure acute concentrations would not result in exceedances of the CS.

measured in surface water and lacks a numeric WQSs in rule. Also, when warranted and described in later sections, these methods will describe how to address mixtures and environmental degradates.

C. Statutory authority for water quality standards

Minnesota's water quality rules and standards are based on both State and Federal requirements and authorities. The Minnesota Pollution Control Agency's (MPCA's) authority to adopt water quality standards and to classify waters of the state by beneficial uses is found in Minn. Stat. § 115.03, particularly subdivisions 1(b) and 1(c). Subdivision 1(b) authorizes MPCA to classify waters, while subdivision 1(c) authorizes MPCA:

> To establish and alter such reasonable pollution standards for any waters of the state in relation to the public use to which they are or may be put as it shall deem necessary for the purposes of this chapter and, with respect to the pollution of waters of the state, chapter 116;

Additional authority for adopting standards is established under Minn. Stat. § 115.44, Subd. 2 and 4. Subdivision 2 authorizes MPCA to:

> …group the designated waters of the state into classes, and adopt classifications and standards of purity and quality therefore. …

Subdivision 4 authorizes MPCA to:

> …adopt and design standards of quality and purity for each classification necessary for the public use or benefit contemplated by the classification. The standards shall prescribe what qualities and properties of water indicate a polluted condition of the waters of the state which is actually or potentially deleterious, harmful, detrimental, or injurious to the public health, safety, or welfare; to terrestrial or aquatic life or to its growth and propagation; or to the use of the waters for domestic, commercial and industrial, agricultural, recreational, or other reasonable purposes, with respect to the various classes established…

Minnesota R. ch. 7050, *Waters of the State: Water Quality Standards for Protection of Waters of the State* implements the requirements of Minn. Statute 115 and has evolved as new water quality programs and issues have emerged and rule language was needed to implement them. In many cases additions were in response to Federal requirements under the CWA. The 1972 passage of the CWA required significant additions to Minnesota's WQSs and rules, primarily in terms of showing consistency in water quality protection goals[9] and implementing point-source controls in the National Pollutant Discharge Elimination System (NPDES) program. As another example, Congress amended the CWA in 1987 to require all states to adopt standards for toxic pollutants. MPCA complied with that requirement with comprehensive amendments in 1990. Minnesota's *Lake Superior Basin Water Quality Standards* in Minn. R. ch. 7052 were adopted in response to 1990 amendments to the CWA, known as the *Great Lakes Critical Programs Act*.

[9] CWA goals: 1) wherever attainable, achieve a level of water quality that provides for the protection and propagation of fish, shellfish, and wildlife, and for recreation in and on the water, and take into consideration the use and value of public water supplies, and agricultural, industrial, and other purposes, including navigation (sections 101(a)(2) and 303(c) of the Act); 2) and restore and maintain the chemical, physical, and biological integrity of the Nation's waters (section 101(a)).

Besides federally mandated rule requirements, CWA Section 304(a) requires that the U. S. Environmental Protection Agency (EPA), administrator of the act, develop and publish *Ambient Water Quality Criteria* (AWQC) for surface water contaminants; these documents provide methods and numeric criteria for protection of aquatic life and human-use of aquatic life (e.g., fish consumption) and water for potable use. These EPA criteria have only guidance status; states and tribes use EPA AWQC as one source of scientifically defensible information to develop and adopt legally enforceable WQSs into rules.

The authority, application, and numeric values promulgated through rulemaking result in pollutant specific WQSs listed in Minn. R. 7050.0220 to 7050.0227 and for Lake Superior WQSs in Minn. R. 7052.0100. When a pollutant lacking a listed WQS is found in surface waters, both rules authorize MPCA to develop numeric site-specific criteria; the criteria are based on the methods for a FAV, maximum criterion (MC) and chronic criteria (CC) described in Minn. R. 7050.0217 and 7050.0218[10] and Minn. R. 7052.0110. On a site basis, for example at remediation sites, the criteria have the same regulatory applications as listed WQSs.

The federal CWA, § 303 (c) (1), also requires the states and authorized tribes to review and amend as appropriate their WQSs every three years. EPA must approve of a state's WQSs to ensure they meet the CWA.

D. Implementation of Water Quality Standards

Water quality standards serve as the foundation for protecting Minnesota's water resources. Besides protecting the designated beneficial uses of groundwater and surface waters, the water quality standards are used to:

1. Assess the quality of the State's water resources.
2. Identify waters that do not meet standards and are polluted or impaired.

MPCA has to assess the water quality of rivers, streams, wetlands, and lakes in Minnesota (Code of Federal Regulations, title 40, part 130). Waters determined to not meet WQSs and therefore, do not support beneficial uses, are defined as "impaired". Impaired waters are listed and reported to the citizens of Minnesota and to EPA in the CWA 305(b) report and the CWA 303(d) Impaired Waters List. The identification of waterbodies that do not meet WQSs and support designated beneficial uses is a high profile and required function of WQSs.

While a complete discussion of MPCA's water quality monitoring approach is outside the scope of this document, it is important to overview aspects of toxic pollutant monitoring as it relates to implementing HH-WQS. Available monitoring data for toxic pollutants to assess HH-WQSs come from NPDES monitoring requirements and ambient surface water monitoring programs. MPCA's monitoring data cover many classes of toxic pollutants including: metals, industrial chemicals, and some legacy organochlorine pollutants, such as PCBs. MPCA also participates on the Interagency Fish Contaminant Monitoring Program (FCMP) for studies on mercury, PCBs, and perfluorinated chemicals (PFCs) in fish tissue. Minnesota Department of Agriculture

[10] In the future, this citation will also include 7050.0219.

provides monitoring data to MPCA on pesticides and their degradates. Other monitoring studies by EPA, MPCA, MDH, and United States Geological Survey (USGS) provide additional information on other pollutants. While not all HH-WQSs have recent or ongoing monitoring data MPCA can evaluate other site-specific (e.g., St. Louis Area of Concern), regional and national, and related information to determine what pollutants need further study and management. This information can be used to help identify future needs for monitoring, as well as, WQS development (as not all pollutant monitored have HH-WQSs). However, not all HH-WQS have sufficient data to fully characterize environmental occurrence based on limitations in analytical methods, scientific understanding, and available resources. MPCA strives to identify and minimize these gaps.

It is also import to note, while HH-WQS include drinking water as a beneficial use of surface waters, drinking water protection is regulated by MDH under the Safe Drinking Water Act (SDWA). Ambient surface water monitoring is not designed to ensure raw or ambient surface water is "drinkable". Regulated drinking water is defined by the SDWA and centers on effectively treating and monitoring of "tap" or finished water. SDWA systems include community, noncommunity, and transient systems, but not individual consumption from untreated surface waters. As stated in the Class 1 narrative (subclasses 1B and 1C), ambient surface water is not intended to be used for drinking without some level of treatment.

3. Set effluent limits and treatment requirements for discharge permits and cleanup activities.

Another important role of WQSs is as the basis for effluent limits. Effluent limits are specified in wastewater dischargers' National Pollutant Discharge Elimination System (NPDES) or State Disposal System (SDS) permits, and define the allowable concentration and mass (e.g., kilograms per day) of pollutants that can be discharged to the receiving water. Effluent limits consider technology, but have to ensure WQS are met. Adoption of the revised human health methods will allow for use in the development of site-specific criteria and future promulgation of pollutant-specific HH-WQS into rule. Updated criteria or standards would then be the basis for Effluent Limits according to relevant procedures.

While implementation of HH-WQSs plays a central role in protecting the beneficial uses of Minnesota's surface waters, limitations exist in this regard. Limitations in monitoring data, analytical methods, and multimedia and nonpoint sources of toxic pollutants can mean that not all pollutants have control measures to achieve the WQSs in all waterbodies. For example, there are pollutants that affect fish consumption that originate from global air sources (e.g., mercury and PCBs), where point source water quality control measure are not going to fully address their impact on water resources. MPCA assesses many pollutants with WQS and if they don't meet WQS list those waterbodies as impaired and develop studies (Total Maximum Daily Load or TMDL) to determine sources and reductions needed to meet WQS. The processes of identifying and listing impaired waters and completing TMDLs are required under the CWA.

Both broader approaches for pollution control and supplemental programs are needed to protect human health, most importantly, MDH's Fish Contaminant Advice (FCA). The Federal Safe Drinking Water Act (SDWA), well codes, and Health Risk Limits (HRLs) administered by MDH provide the foundation for much of Minnesota's drinking water protection. More broadly in groundwater, private well owners also have an important role in testing for drinking water quality. MPCA's HH-WQS[11] consider and develop health-protective drinking water standards that supplement these goals, but are not fully monitored in ambient waters for this use. MPCA continually works with many state, federal, academic, and local partners, and the public to identify and eliminate impaired water and gaps in monitoring data and control measure to ensure HH-WQSs are being met.

[11] In the context of this TSD, HH-WQS refer to Class 2 Chronic Standards and Criteria applied to surface water. MPCA's WQS also include Class 1, Domestic Consumption (DC), drinking water standards. These standards are the SDWA standards incorporated by reference for groundwater and surface waters protected as a source of drinking water.

II. History of Human Health Protection in Minnesota's Water Quality Standards

A. Federal drinking water standards (Class 1)

The focus of this guidance is Class 2 human health-based water quality standards (HH-WQSs). For context on human health protection in WQSs, information is also discussed for Class 1 Domestic Consumption (DC) standards. The Federal *Public Health Service Drinking Water Standards* were incorporated by reference in Minnesota's first water quality rules, predecessors to Minn. R. ch. 7050 (WPC "water pollution control" 1 adopted in 1963 and 14 & 15 adopted in 1967). The concept and methods behind developing numeric standards to protect human health from microbiological, synthetic and elemental pollutants stemmed from methods used in the federal drinking water programs: from the Public Health Service (1962) to EPA's current *Safe Drinking Water Act's Primary and Secondary Drinking Water Standards* (2006). In Minn. R. ch. 7050, these standards are incorporated by reference from 40 CFR 141 as Class 1 DC standards[12].

The basic premise behind federal drinking water standards was to estimate the dose at which a pollutant caused an adverse effect or posed a cancer risk (toxicological evaluation) and the amount of drinking water being consumed by the typical adult (exposure evaluation). By combining these estimates of "risk" and exposure, a health protective standard or advisory level could be derived. Final Safe Drinking Water Act (SDWA) standards (Maximum Contaminant Levels or MCLs) may be based on technology limits for removal of pollutants at drinking water treatment facilities, analytical capabilities, and overall cost and benefits analyses; whereas HH-WQSs, EPA's Office of Waters' *Ambient Water Quality Criteria* (AWQC), and Health Advisory (HA) levels used as guidance in SDWA programs are calculated strictly on human health methods (USEPA 2006a).

B. Fish consumption, recreation, and drinking water uses (Class 2)

Methods specific to developing HH-WQSs to protect the beneficial uses of fish and drinking water consumption and recreation were first adopted in 1990 in Minn. R. ch. 7050 for Class 2 designated surface waters. The protections for human health are an important component for Class 2 designated surface waters. The 1990 human health-based methods have remained largely unchanged ever since. The focus of this guidance is the methods used to develop numeric HH-WQSs for protection of human health historically and proposed for revision and adoption. Numeric HH-WQSs are adopted into rule for a specific Class 2 subclass as chronic

[12] 7050.0410, subp. 2: Class 1 waters, domestic consumption. Domestic consumption includes all waters of the state that are or may be used as a source of supply for drinking, culinary or food processing use, or other domestic purposes and for which quality control is or may be necessary to protect the public health, safety, or welfare.

standards (CSs) or are used as site-specific chronic criteria (CCs) for pollutants measured in surface water, but are lacking listed WQSs (Minn. R. 7050.0217 to 7050.0218 and Minn. R. 7052.0110).

The Class 2 subclasses that correspond to the human health-based beneficial uses are:

- Class 2A (Lake Superior)[13]: Drinking water, fish consumption (lake trout fishery uses 8.5% fish lipid), and recreation;
- Class 2A: Drinking water, fish consumption (cold water aquatic community and trout fishery uses 6% fish lipid), and recreation;
- Class 2Bd: Drinking water, fish consumption (cool-warm water aquatic community uses 1.5% fish lipid-being revised), and recreation; and
- Class 2B (2C and 2D): Fish consumption (cool-warm water aquatic community uses 1.5% fish lipid-being revised) and recreation.

Currently, Minn. R ch. 7050 contains WQSs for 69 toxic substances, 36 are most restrictive for human health (Minn. R. 7050.0222) than aquatic life; Minnesota R. ch. 7052 contains WQSs for 29 pollutants, where for 15, human health is the basis for the most stringent CS (Minn. R. 7052.0100). As stated above, if a CC is needed for a pollutant not among the 69 or 29 listed WQSs, methods are included in both rules to develop site-specific numeric criteria. WQSs promulgated through rulemaking process are primarily based on the same methods described for developing site-specific numeric criteria, but may have also incorporated other scientifically defensible or regional data more appropriately applied on a pollutant-specific basis; the details of pollutant-specific HH-WQSs are found in summary sheets and tables available by request from MPCA. Additionally, complete lists of HH-WQSs and existing site-specific criteria are found in Appendices D1 and D2, respectively.

C. 1990 human health-based algorithms and parameters

HH-WQSs are set at concentrations to protect human users of surface waters. That protection considers the toxicity (deleterious, noxious, or injurious) characteristics of the pollutant and how much a population may be exposed to that pollutant through three designated beneficial uses of surface waters: drinking water, recreational activities, and fish consumption.

In short, HH-WQSs are built on estimating a pollutants' toxicity and a population's potential exposure. Scientific estimates of toxicity and exposure (e.g., use of higher percentile intake rates) and policy directives were integrated to set eight toxicological and exposure parameter[14] values that aim to ensure final WQSs meet acceptable levels of protection from adverse health effects (see full discussion in Appendix B1, EPA 2000b, and Table 1); for example, adult body weight of 70 kg is an average or mean estimate for men and women; whereas, for the fish consumption rate of 30 g/d, MPCA has used an 80th percentile estimate of fish consumption from a population of people that engage in recreational fishing in the Midwest and Great Lakes.

[13] As described in Minn. R. ch. 7052, MPCA has Class 2A WQS specific only to Lake Superior based on the lipid properties of the lake trout. Other Class 2A waters in the Lake Superior basin use the other Class 2A numeric standards.

[14] A "Parameter" is defined as a value used to describe a statistical population (Merriam-Webster).

As explained more fully in EPA's human health guidance for AWQC, the parameters are based on scientifically defensible data with statistical values (means, medians, percentiles, etc.) chosen for broader reasons: consistency in environmental health protection programs across media, soundness of available data, mandated protection level goals, and policy directives (USEPA 2000b).

1990 algorithms used to develop human health-based standards

1. The formula for calculating Class 2A and 2Bd Chronic Standards (CSs) or site-specific Chronic Criteria (CC) for waters protected for the beneficial uses of fish consumption and recreation and as a source for drinking water, for noncarcinogenic chemicals is:

$$CS_{df}{}^{15}, \text{mg/L} = \frac{\text{RfD mg/kg/day x 70 kg x RSC}}{\text{2 L/day + [(0.030 kg/day x BAF L/kg]}}$$

1990 Noncancer Algorithm 1

Where:

CS_{df} = drinking water plus fish consumption chronic standard in mg/L

RfD = reference dose in mg/kg/day

70 kg = standard weight of an adult

RSC = relative source contribution factor

2 L/day = two liters of (drinking) water consumed per day (by adults)

0.030 kg/day = amount of fish assumed to be consumed per day (by people in Minnesota)

BAF = final Bioaccumulation factor in L/kg (liters per kilogram and specific to Class 2 subclass)

2. The formula for calculating Class 2B, 2C and 2D CSs for waters protected for fish consumption and recreation, for noncarcinogenic chemicals is:

$$CS_f, \text{mg/L} = \frac{\text{RfD mg/kg/day x 70 kg x RSC}}{\text{0.01 L/day + [0.030 kg/day x BAF L/kg]}}$$

1990 Noncancer Algorithm 2

Where:
CS_f = fish consumption chronic standard in mg/L
0.01 L/day = assumed incidental ingestion of water,
Other variables as previously identified

[15] The acronyms adopted in 1990, CS_{df} and CS_f, don't directly reference recreational exposure; however, this exposure route is considered in each algorithm. When drinking water intake is part of the algorithm, the 2 L/d is sufficient to address recreational exposure defined as 0.01 L/d. When drinking water isn't part of the algorithm, the incidental water rate is included directly.

3. The formula for calculating Class 2A and 2Bd CSs for waters protected for the beneficial uses of fish consumption and recreation and as a source for drinking water, for carcinogenic chemicals is:

$$CS_{df}\ mg/L = \frac{70\ kg \times 10^{-5}}{q1^* \times [2\ L/day + (0.030\ kg/day \times BAF)]} \quad \text{(1990 Equation C-1)}$$

1990 Cancer Algorithm 1

Where:

10^{-5} = a cancer risk (CR) level of one chance in 100,000.

q1* = the cancer potency factor in days times kg (body weight) per mg (toxicant),

Other variables as previously identified

4. The formula for calculating Class 2B, 2C and 2D CSs for waters protected for fish consumption and recreation, for carcinogenic chemicals is:

$$CS_f\ mg/L = \frac{70\ kg \times 10^{-5}}{q1^* [0.01\ L/day + (0.030\ kg/day)\ (BAF)]} \quad \text{(1990 Equation C-2)}$$

1990 Cancer Algorithm 2

Where:

Variables as previously identified

Table 1: 1990 parameters in human health-based water quality standards

Topic Area	Parameter	Abbreviation	Value and units	Unit description
Toxicological Evaluation- Noncancer or Systemic Toxicant	Reference Dose	RfD	Pollutant specific value mg/kg/d	milligram/kilogram/ day
Toxicological Evaluation- Carcinogens	Cancer Potency Slope	q1*	Pollutant specific value $(mg/kg/d)^{-1}$	one over milligram/kilogram/ day
	Cancer Risk level# (Incremental)	CR	10^{-5}	1 in 100,000
Exposure Evaluation	Drinking Water intake rate-adult	DW	2 L	liters
	Incidental Water intake rate (Swimming Exposure)	IW	0.01 L/d	liters/day
	Fish Consumption-freshwater fish intake rate-adult	FC	0.03 kg/d	kilograms/day
	Parameter	**Abbreviation**	**Value and units**	**Unit description**
	Bioaccumulation Factor	BAF	Pollutant and Use Class specific value L/kg	liters/kilogram
	Relative Source Contribution factor	RSC	Pollutant specific value (0.2 default)	percent
	Body Weight-adult	BW	70 kg	kilogram

For parameter definitions, see Appendix A1

#Used to convey the excess cancer risk target for WQSs for carcinogenic pollutants; not strictly speaking a parameter until applied with the Cancer Potency Slope.

D. 1998 Great Lakes Initiative - human health methods and standards

Concerns about bioaccumulative toxics in the Great Lakes led to amendments to the CWA Section 118(c)(2) in 1990; the *Great Lakes Critical Programs Act* required the US Environmental Protection Agency (EPA) to publish and States to adopt methods and WQS specific to the Great Lakes. MPCA adopted EPA's criteria for this amendment, known as the Great Lakes Initiative (GLI), into State rules in 1998, to address *Bioaccumulative Chemicals of Concern*[16] and persistent chemicals listed in 40 CFR 132. The GLI rule, Minn. R. ch. 7052, applies to the Lake Superior basin. MPCA adopted the GLI methods and pollutant specific WQSs as specified by EPA (USEPA 1995d) and 40 CFR Part 132, Appendices A through D, with a few modifications to reflect State-specific data on fish consumption and fish lipid content; changes to the GLI methods are encouraged to reflect local data, as long as the final methods are "as protective

[16] A Bioaccumulative Chemical of Concern is a pollutant with human health related BAFs greater than 1,000 and meeting the criteria described in Minn. R. 7052.0010, Subp. 4.

as" those published by EPA. The GLI methods reflected some advances in the science of WQSs development, but are fundamentally the same as the 1990 methods in Minn. R. ch. 7050. The human health-based methods are nearly identical as consistency with current state methods was encouraged. However, the bioaccumulation factor (BAF) methods have some important differences from those that were adopted for use statewide in 1990, and provide the basis for the improved BAF methods being proposed in this document and for revision in Minn. Rule ch. 7050 (see Sections IV.C.f., and h. and IV. D).

EPA 304(a) *Ambient Water Quality* Criteria (AWQC) for human health reviewed specifically as part of the GLI provide the toxicological values used for most HH-WQS in Minn. R. ch. 7052 (see Section IV.D.). The GLI methods allowed for a two tier approach for the evaluations depending on the amount of data available for each GLI pollutant. Toxicological values (RfDs or cancer potency) for pollutants with smaller datasets were derived using a "Tier II" method; no comparable method exists in Minn. R. ch. 7050. In the final published EPA AWQC, no criterion was actually derived using the GLI Tier II methods for human health. Subsequently, MPCA's new proposed methods for adoption in both rules will not include the Tier II method. A key reason for not continuing to use the Tier II method is its allowance for very large Uncertainty Factors (UF), up to 30,000, that are generally not acceptable for use now in deriving human health-based standards (MDH 2008a). Another reason stems from the reliance of a single 28-day study, which may or may not be sufficient for developing a criterion or standard. Minimal datasets need now consider the nature of the chemical and more toxicological information than a single-study can typically provide.

E. 2008 mercury fish tissue human health-based water quality standard

MPCA adopted a numeric fish tissue water quality standard to Minn. R. ch. 7050 in 2008. The HH-WQS is based on EPA's *Water Quality Criterion for Protections of Human Health: Methylmercury* (USEPA 2001). The mercury standard is 0.2 milligram of total mercury per kilogram of fish (mg/kg or parts per million, ppm). It applies to total mercury concentrations in edible fish tissue of any species of fish from Minnesota's waters. The promulgation of a 0.2 ppm mercury standard augments the current numeric CSs by providing a more precise level of protection to fish consumers. This mercury HH-WQSs applies directly to the fish medium rather than the water medium.

EPA published the mercury AWQC as a fish tissue concentration rather than a water concentration for many reasons.

- Fish consumption is the primary source of mercury exposure to humans; the standard applies in fish tissue.
- Fish tissue is the medium of interest and concern; therefore, having a fish tissue standard more reliably protects fish consumers by eliminating the need to extrapolate safe mercury levels in fish from water standards by removing the uncertainty in bioaccumulation factors (BAFs).
- There are far more fish tissue data than water data for mercury.

The algorithm for the mercury fish tissue HH-WQS is:

$$CS_{Fish\ Tissue} = \frac{70\ kg\ x\ (RfD - RSC)}{0.030\ kg/day}$$

$CS_{Fish\ Tissue}$ or CS_{ft} = for total mercury of 0.20 mg/kg or ppm (parts-per-million)

Given:

RfD = reference dose in mg/kg-day of 0.0001 mg methylmercury/kg bodyweight-day

70 kg = standard weight of an adult

RSC = relative source contribution factor of

2.7 x 10^{-5} mg methylmercury/kg bodyweight-day

0.030 kg/day = amount of fish assumed to be consumed per day (Minnesota)

The application of HH-WQS in fish tissue for mercury sets the stage for including methods and algorithms for developing fish tissue standards for additional bioaccumulative pollutants. As discussed in more detail in the BAF section (IV.C.g.), the waterbody[17] and aquatic community characteristics directly affect how much a pollutant will bioaccumulate and these characteristics can differ greatly. However, determining and adopting a protective fish tissue-based CS_{ft} is more accurate and meaningful for highly bioaccumulative pollutants, because that is the medium of highest exposure and where more information is available on pollutant concentrations. MPCA is proposing to adopt such methods to include a fish tissue CS (or CC) in addition to water CSs (or CCs) as described in Section IV.E.

[17] The term *waterbody* is used in this document, but is synonymous with *water body* or *body of water* as used in Minn. R. ch. 7050.

III. Methods for Human Health-Based Standards

A. Foundation in federal EPA and state programs

Human health-based water quality standards (HH-WQSs) have their basis in standard algorithms and methods that are refined with pollutant-specific data. The development, adoption, and use of methods for protection of human health for surface water users has centered on the Clean Water Act (CWA) and objectives to reduce *Priority Pollutants* (many defined as such because of toxicity to humans or aquatic life communities)[18]. Section 304(a) of the CWA also specifically directs EPA, as administrator for the Act, to develop methods and pollutant-specific *Ambient Water Quality Criteria* (AWQC) for use by states and tribes to develop these protections through WQSs; EPA has been publishing *Water Quality Criteria* since 1976. As of 2009, EPA had published national AWQC for about 160 pollutants, with over 100 having human health-based criteria, including 10 based on organoleptic factors (taste and order). These criteria formed the basis for many of MPCA's numeric chronic standards (CSs) adopted in 1990 and 1994 in Minn. R. ch. 7050. In 1995 EPA published updated criteria for 29 pollutants for the Great Lakes Initiative (USEPA 1995d); these standards are incorporated by reference (using Minnesota's fish consumption rate and fish lipid values) in Minn. R. ch. 7052.

EPA develops AWQC on a national level with expectations that states and tribes incorporate more representative local data when available when adopting WQSs. MPCA has always incorporated Minnesota-specific data in HH-WQS. Local data include fish consumption rates and fish lipid values developed by MPCA in 1990, and toxicological values, Relative Source Contribution factors (RSCs), and Cancer Risk levels (CRs) developed by the Minnesota Department of Health (MDH) in their groundwater protection program (MPCA 1989; Appendices B1-B3).

MDH develops and adopts *Health Risk Limits* (HRLs) for groundwater to protect human health for drinking water use under the authority of Minnesota's *Groundwater Protection Act* (Minnesota Statutes Sections 103H.001 *et. seq.*); the HRLs are promulgated in Minn. R. ch. 4717. HRLs are comparable to HH-WQSs in four important ways:

1. Basis in human health protection only;
2. Methods build on EPA risk assessment guidance, methods, and datasets used in the Office of Waters' Safe Drinking Water and Clean Water Act programs and the Office of Pesticide Programs;
3. Incorporate Minnesota specific Legislative mandates relevant to infant and children protections (the focus of these revisions); and
4. Put into practice the latest risk assessment methods and pollutant-specific toxicological evaluations (2009 and 2011 revisions to Minn. R. ch. 4717).

[18] The 1972 CWA lists 172 *Priority Pollutants* that were deemed a concern in the nation's waters at that time. EPA, states and tribes were required to address these pollutants in water quality standards (http://www.epa.gov/waterscience/criteria/wqctable/).

MPCA's policy is to maintain consistency between HH-WQSs and MDH's HRLs and other MDH risk assessment protocols to comply with the intent discussed in the 1989 MPCA SONAR and to recognize MDH's roles as the lead state agency for toxicological evaluations and consultations; MDH provides risk assessment support to MPCA's Air and Remediation programs and the Minnesota Department of Agriculture's Remediation and Pesticide programs (See http://www.health.state.mn.us/divs/eh/risk/programs/index.html).

B. Protection level goals

It is the intent of MPCA to fully protect the waters of the state such that they are suitable for fisheries and all designated beneficial uses consistent with available toxicity information and ambient water quality data and rules.

Minn. R. 7050.0150, subp. 3, contains the narrative requirements that form the basis for protection level goals for Class 2 HH-WQSs for the beneficial uses of drinking water, fish consumption, and recreation.

> Subp. 3. **Narrative standards.** For all Class 2 waters, the aquatic habitat, which includes the waters of the state and stream bed, shall not be degraded in any material manner, there shall be no material increase in undesirable slime growths or aquatic plants, including algae, nor shall there be any significant increase in harmful pesticide or other residues in the waters, sediments, and aquatic flora and fauna; the normal fishery and lower aquatic biota upon which it is dependent and the use thereof shall not be seriously impaired or endangered, the species composition shall not be altered materially, and the propagation or migration of the fish and other biota normally present shall not be prevented or hindered by the discharge of any sewage, industrial waste, or other wastes to the waters.

Objectives of the narrative standard for human health protection are further described in Minn. R. 7050.0217, subp. 2.

> Protection of human consumers of fish, other edible aquatic organisms, and water for drinking from surface waters means that exposure from noncarcinogenic chemicals shall be below levels expected to produce known adverse effects; and the incremental cancer risk from exposure to carcinogenic chemicals, singly or in mixtures, shall not exceed one in 100,000. The combined risk from mixtures of carcinogens will be determined as described in part 7050.0222, subpart 7, item D.

Fully incorporated into the narrative standards is achievement of the CWA goal of "fishable" surface waters which means, not only protecting aquatic life from toxic effects, but also ensuring edible fish should not have levels of toxic chemicals in their tissue that make them unsafe for humans to eat. Also, edible aquatic organisms should not have levels of chemicals in their tissues that impart unacceptable tastes to the flesh (organoleptic effects)[19]. Thus, chronic

[19] While EPA's 2000 guidance states that EPA will no longer develop organoleptic *Water Quality Criteria*, MPCA will retain those WQSs on a pollutant-specific basis as they were more stringent than needed for human health protection or until EPA publishes updated criteria.

standards (CSs) are based on human health concerns if the concentration of a chemical needed to protect human consumers of freshwater fish is lower than the concentration needed to protect the organisms themselves from the toxic effects of a chemical.

For pollutants that pose health risk through their bioaccumulation in fish tissue, Minnesota also has a narrative standard in Minn. R. 7050.0150, subp. 7 that limits fish tissue contaminants to levels that allow safe consumption of fish as often as one meal per week. The original narrative standard ("*…nor shall there be any significant increase in harmful pesticide or other residues in the waters, sediments, and aquatic flora and fauna…*") dates to the first statewide water quality rule in 1967. In a 2003 rulemaking, MPCA linked the level of contaminants that are acceptable and unacceptable in fish to MDH Fish Consumption Advice by expanding on the original narrative standard in Minn. R. 7050.0150, subp. 7.

> **Impairments of waters relating to fish for human consumption**. In evaluating whether the narrative standards in subpart 3, which prevent harmful pesticide or other residues in aquatic flora and fauna, are being met, the commissioner will use the residue levels in fish muscle tissue established by the Minnesota Department of Health to identify surface waters supporting fish for which the Minnesota Department of Health recommends a reduced frequency of fish consumption for the protection of public health. A water body will be considered impaired when the recommended consumption frequency is less than one meal per week, such as one meal per month, for any member of the population. That is, a water body will not be considered impaired if the recommended consumption frequency is one meal per week or any less restrictive recommendation such as two meals per week, for all members of the population. The impaired condition must be supported with measured data on the contaminant levels in the indigenous fish.

The human health protection required by the CWA also includes "recreation in and on the water", many times shortened to "swimmable." MPCA Class 2 HH-WQS address this use through an incidental water intake rate and separately in the microbiological or bacteriological standards for *Escherichia coli* (not addressed in this guidance).

To implement the CWA protection goals, EPA published guidance, methods, and numeric criteria beginning in the 1970s to set the pollutant-specific concentrations in surface waters that would protect surface water uses of drinking water, fish consumption, and recreation. As explained in detail in following sections, MPCA adopted EPA guidance (mostly updated in 1980) with Minnesota-specific parameters and algorithms in 1990 to develop and promulgate numeric HH-WQSs. These numeric CSs were originally, and for the most part remain, expressed as water concentrations.

MPCA augmented the narrative and numeric water standards in 2008 by adopting the first fish tissue numeric HH-WQSs for mercury (for details see Section II.E.). The standard was based on EPA's 2001 AWQC for methylmercury. The additional fish tissue HH-WQS supplements protection from bioaccumulative pollutants, because fish consumption is the main route of exposure and better media to monitor and assess exposure.

C. Chronic standards: basis in "lifetime" of protection

HH-WQSs are developed to provide protection, not only for the general population over a lifetime of exposure (average life expectancy of 70 years used in risk assessment), but also for special subpopulations or distinct life stages of a person's life where, because of higher than average exposure rates (high water or fish consumption), or because of biological sensitivities, have a higher risk of adverse effects. Consistent with recent policy statements, guidance, and rule revisions from MDH and EPA (Minnesota Department of Health (MDH) and Agency for Toxic Substances and Disease Registry 2006, MDH 2008a), it is MPCA's goal as well, to more closely assess and protect developmental life stages from preconception through adolescence by using life stage specific approaches with respect to exposure parameters, or through the application of more accurate pollutant-specific data in place of default parameters when sufficient data are available. While, developmental life stages are not the only life stages with possibly higher or unique risk, the scientific basis is more complete and defined for developmental responses than at other distinct times in a person's lifecycle (e.g., aged adults or health-compromised individuals) (USEPA 2006b). Incorporating more protections for developmental life stages though will mean reducing exposure to pollutants for all surface water users, as their higher intake rates generally result in more stringent CSs.

While the goal of protection for a lifetime, including all life stages of a human, is not new, the risk assessment methods to accomplish this goal have substantially improved since 1990; however, even with the more advanced risk assessment methods and comprehensive toxicological evaluations, the original protection objectives adopted in 1990 for human health-based CSs are still appropriate and meaningful.

As discussed throughout this guidance, HH-WQSs are intended to provide the same level of protection as MDH HRLs. This protection level goal for linear carcinogenic chemicals is a lifetime risk no greater than one additional case of cancer in a population of 100,000 (Minn. R. 7050.0218, subp. 6.B and Minn. R. 4717.7820, subp. 4). The protection level goal for noncarcinogenic[20] toxic pollutants is to achieve an exposure level that is without an appreciable risk of adverse effects, including sensitive life stages or subpopulations as defined by the Reference Dose (Minn. R. 7050.0218, subp. 2.Z. and Minn. R. 4717.7820, subp. 20).

The HH-WQSs are implemented by comparing the CSs with 30-day average surface water concentrations, in contrast to the 4-day averaging time for aquatic toxicity-based CSs. Originally, the longer averaging period for HH-WQSs were adopted to reflect the very long, lifetime protection equating to "70 years" of potential exposure to the pollutant and the general duration for many pollutants to reach steady-state in fish tissue. With the additional consideration of less-than-chronic health effects provided by MDH's new acute, short-term, and subchronic RfDs and drinking water intake rates and proposed application in HH-WQSs (see Section V.B.c.), the 30-day duration also matches the short-term duration and provides meaningful protection for short-term and chronic durations of exposure. If appropriate, a one-day averaging time would be used if a HH-WQS based on the acute duration toxicity resulted in the most or equally stringent value for a pollutant-specific HH-WQS (see Section VII.B.).

[20] The term *Noncarcinogenic* is used in Minn. R. ch. 7050 and is synonymous with the terms *Noncancer* and *Systemic Toxicant*. *Nonlinear carcinogens* when addressed with a RfD are also included in this goal.

IV. Revised Methods for Human Health-based Water Quality Standards Amendments

The proposed revisions to methods and algorithms for HH-WQSs are built on consistency with MDH and EPA protections for drinking water uses and more broadly surface water users. The revisions include both scientifically defensible and policy based improvements as described fully in the following sections under *Defining Developmental Protection, Toxicological Evaluations, Exposure Evaluations, and Revised Parameters and Algorithms for Human Health-based Water Quality Standards.*

A. Defining developmental protection—A key revision guiding principle

a. Introduction

A need to better assess and protect infants and children from environmental hazards has clearly been demonstrated in risk assessment methods, policy, and legislation. Addressing vulnerabilities to infants and children really translates into the need to better address all developmental life stages: preconception through adolescence (USEPA 2005c). *Lifestages* as defined in this document are based on EPA's recent approaches to standardizing developmental protection in risk assessments:

> "Temporal stages of life that have distinct anatomical, physiological, and behavioral or functional characteristics that contribute to potential differences in vulnerability to environmental exposures. A lifestage approach to risk assessment considers the relevant periods of exposure in developmental lifestages and subsequent outcomes that may not be expressed until later lifestages. This approach explicitly considers existing data as well as data gaps for both exposure and health outcomes at various lifestages." (USEPA 2006b)

In 2005 EPA published guidance that distinctly identifies *developmental lifestages*[21] by age groups that need consideration in risk assessments. The age groups start prior to conception and continue until adulthood (USEPA 2006b):

[21] EPA's 2005 guidance defined and used "life stage(s)" as a single word. However, in later guidance, reverted back to the more widely used convention of two words. When directly referencing the 2005 guidance, this TSD uses "lifestage(s)' as one word, otherwise, it is used in the more conventional way as two words.

Table 2: Development lifestages and age groups from USEPA 2006b, Table 3-1

Lifestages	Age Groups[a]
Preconception	Reproductive age adult
Prenatal	Conception to birth
Infant	Birth to <1 month
	1-<3 months
	3 to <6 months
	6 to <12 ninths
Child	1 to <2 years
	2 to <3 years
	3 to <6 years
	6 to <11 years
Adolescent	11 to <16 years
	16 to < 18 years
	18 to <20 years

[a] The age groupings from birth to adulthood are from U.S. EPA (2005c). These standard age groups were developed based on the results of a peer involvement workshop focused on developmental changes in behavior and physiology impacting exposures to children.

For the purposes of better refining the developmental life stage assessment, in addition to EPA's Age Groups, three additional life stage characteristics are important to HH-WQS. Two definitions from the World Health Organization (International Programme on Chemical Safety (IPCS) 2006) are *neonates* as infants from birth to 28 days of age (or < 1 month) and *adolescents* as the ages "beginning with the appearance of secondary sexual characteristics to achievement of full maturity", so not a specific age range, but generally referring to ages 11 to 21 (EPA) or 12 to 18 (WHO). The third relates to *reproductive age adults*; in HH-WQSs reproductive age women who are or may become pregnant are an important subpopulation because of the recognition that fish consumption can be a primary source of neurodevelopmental toxicants (i.e., mercury and PCBs). EPA defines this subpopulation as women ages 15 to 44 (USEPA 2000b).

Three key aspects to consider when assessing developmental protections are:

1) Starting from before fetal development and through adolescence there can be stages of greater sensitivities to the toxic effects of environmental pollutants, and the potential for higher exposure rates than during other life stages.
2) Lifetime or chronic protection has to incorporate additional measures to address adverse effects that can occur over less-than-chronic exposure durations; this includes both developmental and nondevelopmental life stages.
3) Adverse changes occurring during a developmental life stage may not be manifested until a later life stage.

MDH further defined and implemented these important points in their new methods for Health Risk Limits (HRLs) (MDH 2008a)[22]:

> "For the purpose of this HRL revision, the term "development" is used to refer to the broad range of effects that occur as a result of exposure during periods when cells or tissues undergo differentiation, rapid replication, and maturation. Thus, adverse developmental effects can result from exposure of either parent prior to conception, exposure of the mother during gestation, exposure of the breastfeeding mother or infant, or exposure during childhood through the time of sexual maturation. For chemicals that cause developmental effects, the revised HRLs provide information about the organ or system affected, for example, "development (skeletal)" or "development (cardiovascular)." Developmental effects may be expressed and detected long after the damage was initiated and long after the damaging exposure occurred."

b. Windows of developmental toxicity

As introduced in the first key aspect, toxicity and exposure are not constant across developmental life stages; upon closer examination of developmental toxicity, sensitivity—while it can depend on organ or system targeted by a given pollutant—is typically highest during the prenatal life stage. Illustrated in the following table (USEPA 2003a), critical developmental windows are pronounced in the prenatal life stage and stem from the fact that processes initiated prenatally set the stage for differentiation and maturation of an organism's biological systems. Early developmental processes build on a series of cascading, tightly orchestrated steps needed for normal structure and function; subsequently, disruption to early developmental processes can result in major malformations and physical deformations[23] or less severe or obvious deficits that result in reduced functions (USEPA 2003a). This is a primary reason for MDH implementing acute and short-term reference doses (RfDs) to address prenatal and neonatal developmental effects discussed in detail later in this document.

[22] MDH does not specifically define "life stages" in Minn. R. 4717; however, the accompanying SONAR to the 2009 revisions discusses "life stage" evaluations as particularly critical to examining developmental toxicity and overall their new methods are founded on these principles (MDH 2008).

[23] Note: Malformations and physical deformations are very apparent outcomes from toxicology laboratory animal assays and in a few cases, actual human events were linked to pharmaceuticals (thalidomide) or environmental pollutants (dioxin), and had served historically as the basis for assessing health effects from toxic pollutants (e.g. Minn. Statute 115). The 2001 Health Standards Statute (Minn. Stat. §144.0751) reflects a more current understanding of the full suite of possible adverse outcomes associated with toxic pollutants.

Table 3: Stages of prenatal and postnatal organ structural development from USEPA 2003, Table 1

Organ System	Early Prenatal	Mid-Late Prenatal	Postnatal
Central nervous system	3 – 16 weeks	17 – 40 weeks	Continues into adulthood
Ear	4 – 16 weeks	17 – 20 weeks	--
Heart	3 – 8 weeks	--	--
Immune system	8 – 16 weeks	17 – 40 weeks	Immunocompetence: 0 -1+ years immune memory: 1- 18 years
Kidneys	4 – 16 weeks	17 – 40 weeks	Nephrons mature in outer cortical region, providing ability to concentrate urine.
Limbs	4 – 8 weeks	--	--
Lungs	3 – 16 weeks	17 – 40 weeks	> 80% of alveoli are formed after birth to age 8-10
Palate	6 –10 weeks	--	--
Reproductive system	7 – 9 weeks	10 – 40 weeks	Sexual maturation, breast, and cervix development: 9-16 years
Skeleton	1 – 12 weeks	--	--
Teeth	12 – 16 weeks	17 – 24+ weeks	Primary dentition: 4 months after conception to 3 years postnatal Permanent dentition: 3 months after birth to 25 years.

Developing less-than chronic RfDs based on developmental-specific or any adverse effect recognizes that chronic[24] exposure is not always required to elicit toxic effects. Evaluations of toxicological studies that use stop-start and interim time point evaluations have shown that exposures less than 10% of an organism's life span can manifest toxic responses (MDH 2008a). For many pollutants, these acute to subchronic adverse effect levels and RfDs may occur at dose higher than the ones leading to chronic effects; however, that is not always the case. And an equally important aspect discussed below, is that when less-than-chronic RfDs are applied with shorter duration exposure rates that are often higher than chronic rates, the final protective concentrations for shorter durations are more stringent than the chronic.

[24] MPCA plans to adopt MDH's definition of *chronic* as a period of time greater than approximately 10% of the life span in humans (70 years) and applied in exposure rates as durations 8 years or longer (MDH 2008a).

HH-WQSs will evaluate and address adverse effects to developing systems as part of the noncancer toxicological evaluation, including detailed review of developmental toxicity data and consideration of other health effects that can occur at less-than-chronic durations (see details in Section IV.B. *Noncancer Evaluations*).

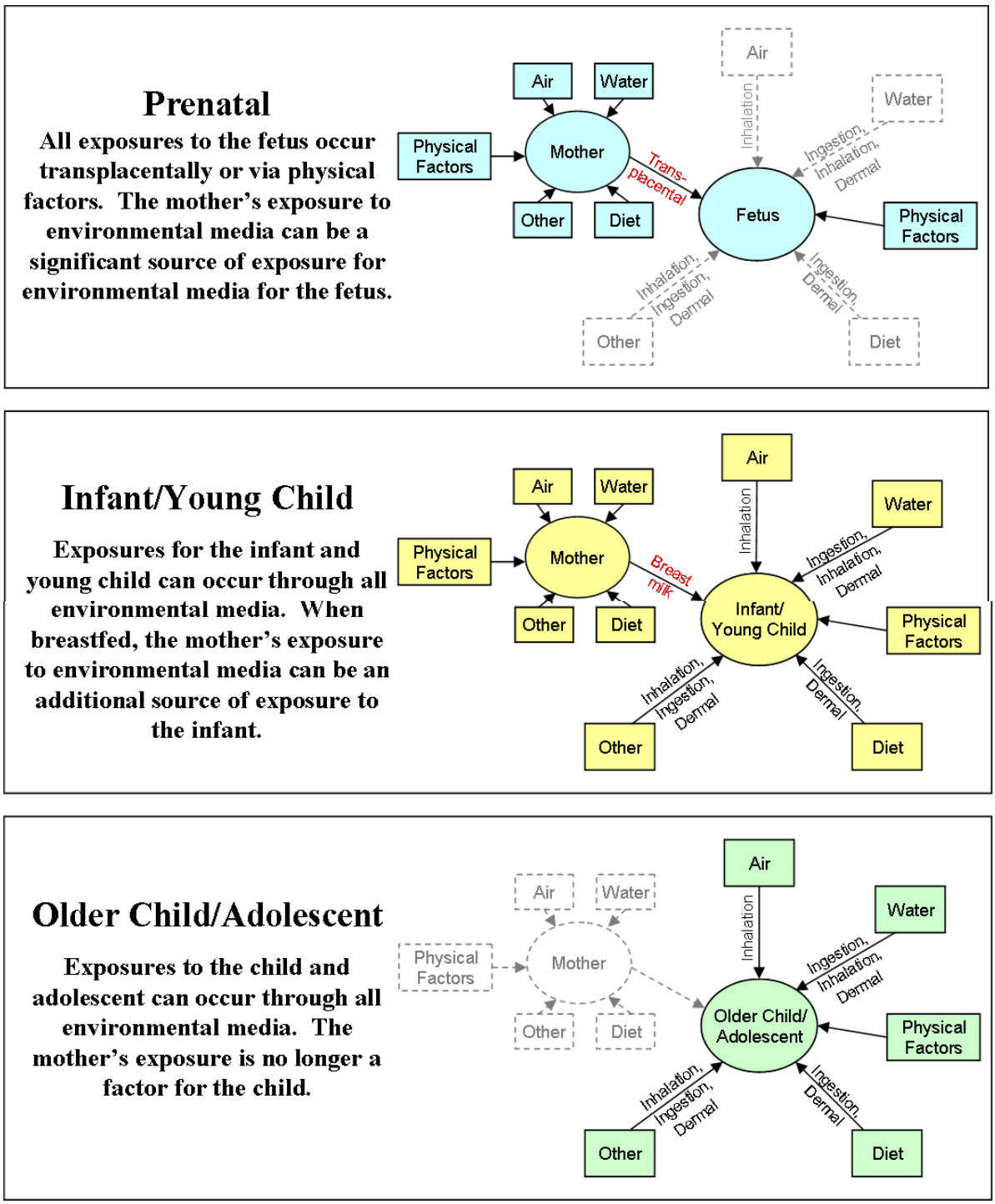
Recent reviews of cancer processes and incidence have also shown the developmental life stage to be more susceptible. As referenced above, developing systems are characterized by rapid cell differentiation and growth coupled with less mature natural repair mechanisms to protect from alterations to genetic material, both factors lead to higher sensitivity to cancer-causing processes. Available data reviewed by EPA and MDH have pointed to the sensitivity to carcinogens to be highest from birth up to age two, with another window of relatively higher sensitivity between ages two to 16 (USEPA 2005b, MDH 2008a) (see IV.B. *Cancer Evaluations*).

c. Exposure differences

The other important component of developmental protection is characterizing how exposure (meaning contact and intake of a pollutant) changes throughout the developmental life stages and differs from that of adults. Details are covered as part of the *Exposure Evaluations,* Sections IV. C.), but in short, assessment of exposure across developmental life stages has to examine:

- Routes a pollutant is taken in, which vary significantly during developmental life stages from prenatal, transplacental exposure that is based on the pregnant mother's exposure, to direct ingestion (see Figure 1 from USEPA 2006b);
- Profiles of fluid and nutrition sources that incorporate important differences from adults: beginning with intake of breast milk and formula to addition of solid foods and reliance on fewer food groups (e.g., apples and dairy products) (USEPA 2008);
- Magnitude on a body weight basis, which is generally higher for infants and children than adults; and
- Fate in an organism (described by toxicokinetics), including very different metabolic and detoxifying processes (USEPA 2005c).

Figure 1: Exposure routes during developmental life stages from USEPA 2006b, Figure 4-9



Exposure profiles for HH-WQS need to consider pollutants in surface waters that may be ingested through drinking water, incidentally when swimming (or other recreational activities), and fish consumption. MDH has fully characterized exposure for developmental protection for drinking water uses (MDH 2008a); their review found that the neonatal life stage is the highest exposure when formula prepared from drinking water is the primary source of fluid and nutrients. The profile for incidental water ingestion is different: swimming in pools is noted for infants in EPA's Exposure Factors Handbooks (USEPA 2008, USEPA 2011), but generally wading and swimming in lakes and rivers would not start until an older age and would be supervised to limit ingestion of water. Infants would also generally not start eating fish, at the earliest, until around six months of age when solid foods are being incorporated into their diet. However, developmental exposure from pollutants in fish can occur prenatally and postnatally based on a pregnant or nursing mother's fish intake. These important differences in sources and routes of pollutant exposure are critical to setting appropriately protective water quality standards for developmental life stages.

d. Latent effects

The third issue raised in regards to developmental protection is that perturbations or alterations to developing systems may not be evident until a later life stage, which is especially relevant to cancer processes (USEPA 2006b). Therefore, assessment of developmental toxicity has to consider potential latent effects. More detailed information on a pollutant's modes or mechanisms of action lend evidence in both noncancer and cancer characterizations for assessing latent effects. Many pollutants have at least relevant mode of action (MOA) data to postulate possible latent effects in cases where available laboratory animal models are not sufficient to assess directly (CA OEHHA 2005). Measures of reproductive toxicity, particularly when acting through endocrine MOAs, are other important factors to consider for fully accounting for latent effects. MDH uses the full complement of toxicological data, including knowledge of MOA and mechanisms of action to address latent events in development of RfDs and cancer potency slopes (MDH 2008a).

e. Conclusions and application by MDH HRL rule and MPCA proposed HH-WQSs

MDH provides a very thorough discussion on the critical aspects to evaluating developmental protection for a pollutant in their supporting Statement of Need and Reasonableness (SONAR) for the recent HRL rule amendments (MDH 2008a), implemented in Minn. R. ch. 4717. Key aspects to ensuring vulnerable developmental life stages, and subsequently infants and children, are protected include:

- Fully reviewing a pollutant's toxicological profile to cover short duration adverse effects, particularly during developmental life stages.
- Identifying data gaps for these effects (developmental and reproductive) that may require use of a database uncertainty factor (UF) in the development of acute or short-term reference doses (RfDs).

- Accounting for known and potentially higher susceptibility in early life stages to carcinogens through use of the Age Dependent Adjustment Factors (ADAF) or chemical-specific lifetime adjustment for cancer potency ($AF_{Lifetime}$).
- Application of higher drinking water intake rates by duration and on a per body weight basis to better reflect exposure during windows of susceptibility for adverse noncancer effects and greater cancer risk.
- Use of relative source contribution factors (RSC) that are more appropriate for the new exposure durations as recommended by EPA's RSC Exposure Decision Tree (2000b).

For HH-WQSs many of the same considerations are warranted and expanded as the exposure profile focuses on toxic pollutants in surface waters, which besides drinking water uses in some waters, also considers fish consumption and recreation (e.g., swimming) in all waters. HH-WQSs are based on setting a chronic standard for long-term or lifetime protection; however, if pollutant-specific data demonstrate that a less-than-chronic duration is more appropriately applied to ensure developmental protection, the methods discuss the process by which the data will be used. Specifically, revisions to human health methods will include:

- A process for when HH-WQSs would be based on less-than-chronic durations when vulnerability (both in toxic effects and exposure) is demonstrated for developmental life stages and these protections are more stringent than those based on chronic effects.
- Applying ADAF or other pollutant-specific lifetime cancer potency adjustment factors using the same protocol as MDH's HRL rule.
- Use of revised drinking water intake rates as defined by MDH and fish consumption and incidental water intake rates as developed by MPCA that when incorporated into the new and revised algorithms will improve HH-WQSs protection for all life stages.
- Review and application of RSC values as recommended by EPA's RSC Exposure Decision Tree (2000b).

B. Toxicological evaluation

a. Introduction

The *Toxicological Evaluation* for HH-WQSs examines and defines adverse health outcomes in both qualitative and quantitative terms to determine each pollutant's key toxicological endpoints. Qualitative descriptions of a pollutant's toxicological profile begin with examining all available data to distinguish noncancer from cancer processes and adverse effects. Toxicological data primarily come from animal laboratory assays, with refinements from human epidemiological studies if data exist. Quantitatively, data are used to derive reference doses (RfDs) and cancer slope factor (CSF or q1*), and new, early-life cancer ADAFs or pollutant-specific $AF_{Lifetime}$. Historically, MPCA has relied on MDH and EPA for toxicological evaluations for use in developing HH-WQS (MPCA 1989, Minn. R. chs. 7050 and 7052). Many toxicological evaluations for existing HH-WQSs primarily came from EPA's Integrated Risk Information

System (IRIS), Office of Pesticide Program's (Re) registration Eligibility Decisions (REDs), and Office of Water's Safe Drinking Water Program or *Ambient Water Quality Criteria* (AWQC) or were developed by MDH.

EPA has published newer methods to toxicological assessment in the 2000 guidance; however, this guidance and no single guidance from EPA has incorporated all the latest advancements until MDH published updated guidelines and rules for toxicological evaluations in MDH's 2008/2009 *Statement of Need and Reasonableness* (SONAR) and Minn. R. ch. 4717. Chemical-specific evaluations for drinking water standards are available from MDH (see http://www.health.state.mn.us/divs/eh/risk/guidance/gw/) or summarized in Minn. R. 4717.7860. Future HH-WQSs in Minn. R. chs. 7050 and 7052 will be based on MDH methods for toxicological evaluations used to develop Health Risk Levels (HRLs) or health based guidance (e.g., HBVs) as incorporated into the revised methods to be adopted in Minn. R. ch. 7050. The revised methods also meet CWA requirements for WQSs adopted by states. Site-specific Chronic Criteria (CC) would also be based on toxicological values from MDH if available for a pollutant or developed as described in the revised methods for HH-WQS.

To provide context to MPCA's use of the toxicological evaluations, summaries of key topics are discussed in the next two sections: *Noncancer and Cancer Evaluations*. To develop toxicological values, critical studies are identified that meet guidelines for acceptable scientific data and studies. The studies also have to record an "adverse effect", which MDH defines as:

> a biochemical change, functional impairment, or pathologic lesion that affects the performance of the whole organism or reduces an organism's ability to respond to an additional environmental challenge (MDH 2008a).

These studies provide the basis for characterizing noncancer health effects. Studies may provide information on cancer processes and outcomes when designed to detect tumors (neoplasms) or alterations to a cell's genetic expression or products (e.g., genotoxic and mutagenic assays). Of note, as the science of risk assessment has advanced and more is known about the modes of actions in the steps of carcinogenesis, toxicological evaluations are now moving toward describing a pollutant's toxicity under a continuum of responses and not by those two distinct definitions (National Research Council of the National Academies (NRC) 2009).

b. Noncancer evaluations

Technical background

EPA has published and improved guidance dealing with noncancer toxicological evaluations since the HH-WQS algorithms were adopted in 1990. Many of their improved methods have evolved from reports from the National Academies of Science (NAS). Most recently, EPA has moved to specifically add life stage considerations into risk assessment frameworks across their programs (Appendix A3). MDH has used EPA and NAS reviews relating to improving life-stage protections and other recommended improvements to risk assessment, along with their own technical reviews and response to Minnesota policies and legislation, to put into practice

improved health protective standards in their 2009 revisions to the HRL rule (Minn. R. ch. 4717). MDH's supporting SONAR fully explains their new methods and the reasons and scientific data behind them (MDH 2008a).

MDH improved their approach to noncancer toxicological review in two principle ways. MDH has identified a broad suite of noncancer studies that are vital to fully understanding a pollutant's toxicological profile; by cataloging the availability of these types of study results, there is more clarity in the data gaps and their handling of those gaps (e.g., use of an extra data Uncertainty Factor). The categories of studies cover immunotoxicity, reproductive toxicity, endocrine effects, neurotoxicity, general organ system and developmental toxicity. The other improvement, which was also supported by an MDH initiated Expert Review Panel, was to more comprehensively address adverse effects by examining less-than-chronic durations of exposure (chronic being typically defined as greater than 10% of a human's life span and foundation of most risk assessments). EPA had previously recognized in a review of their protocol for developing noncancer toxicological values (reference doses and reference concentrations) that only having these values for use in risk assessment based on chronic exposures missed the fact that some adverse effects can occur after exposures that shorter in duration and occur during windows of higher developmental susceptibility (USEPA 2002b). MDH also determined that to more accurately protect for those shorter duration effects, including developmental toxicity, exposure parameters need to match those same durations and cover the life stages of highest exposure (MDH 2008a).

The algorithm for noncancer HH-WQSs centers on the RfD. Currently, MPCA defines a RfD as "an estimate of a daily exposure to the human population, including sensitive subpopulations, that is likely to be without appreciable risk or deleterious effects over a lifetime; expressed in units of daily dose, mg/kg/day" (Minn. R. chs. 7050 and 7052). MDH has expanded this definition in the HRL rule to include multi-duration RfDs (Minn. R. ch. 4717):

> An estimate of a dose for a given duration to the human population, including susceptible subgroups such as infants, that is likely to be without an appreciable risk of adverse effects during a lifetime. It is derived from a suitable dose level at which there are few or no statistically or biologically significant increases in the frequency or severity of an adverse effect between the dosed population and its associated control group. The RfD includes one or more divisors, applied to the suitable dose level, accounting for: (i) uncertainty in extrapolating from mammalian laboratory animal data to humans; (ii) variation in toxicological sensitivity among individuals in the human population; (iii) uncertainty in extrapolating from effects observed in a short-term study to effects of long-term exposure; (iv) uncertainty in using a study in which health effects were found at all doses tested; and (v) uncertainty associated with deficiencies in the available data. An HRL is not derived if the product of the divisors exceeds 3,000. The RfD is expressed as mg/kg-day.

MDH has also defined the "Durations" that are used to set a maximum of four RfDs (when toxicological data are available) for a chemical:

- Acute (24 hours or less);
- Short-term (repeated exposure for more than 24 hours, up to 30 days);
- Subchronic (more than 30 days, up to approximately 10% of a lifespan in humans); and
- Chronic (more than approximately 10% of the life span in humans).

Based on standard chronic toxicity studies, 70 years is typically used as the human life span, even though newer estimates and some applications use 78 years. Practically translated into an applicable duration for chronic RfD development, studies considered "chronic" equate to a minimum of 10% of a human's life span, defined as eight years by MDH for RfD development. MDH may also have relevant data to determine an alternate chemical-specific duration is more appropriate than the four default durations for use in developing a RfD; that determination is based on both toxicokinetics and toxicodynamics evaluation (MDH 2008a).

MDH identifies relevant studies within each of these durations or chemical-specific duration when warranted, which are then used to develop a Point of Departure (POD) and appropriate Dosimetric Adjustment Factors (DAFs) for calculation of the Human Equivalent Dose (HED)(default being $BW^{3/4}$); all these evaluations are applied with appropriate UFs to calculate the RfDs (MDH 2008a, MDH 2011) . In their toxicological evaluations, MDH also lists the adverse effects that occur at the lowest dose (critical effects) and at slightly higher doses (co-critical effects). The significance of these lists is that MDH draws on these to designate Health Risk Index Endpoints (Health Endpoints) at the organ or system level for use in addressing protection from exposure to multiple chemicals (see application details in SectionV.B.f.). Use of Health Endpoints fits EPA guidance on the recognition of specific MOA or target organ or system information when available for assessing mixtures of pollutants (MDH 2008a). The Health Endpoints also make apparent the adverse effects of greatest concern.

Within this context, developmental toxicity, if noted, will principally be addressed as part of the datasets used to develop acute and short-term RfDs. Studies designed to examine developmental effects, which can also fall under the categories of neurological and reproductive toxicity, use short duration exposures and center on identifying adverse outcomes during fetal and neonatal life stages (MDH 2008a). As discussed in Section IV.A., *Defining Developmental Protection,* MDH has clearly described the life stages and toxicological effects that are considered under the Health Endpoints of "Developmental Toxicity" (Minn. R. 4717.7820 C. subp. 8). These key effects considered relevant to developmental toxicity are: death of a developing organism; structural abnormalities; altered growth; and functional deficiencies. Health Endpoints for developmental effects are handled somewhat differently. MDH described the importance for assessing any toxic impact to a developing human more broadly as a process than by common organ or system effects.

In addition, MDH broadly considers toxicity data on any health effect that demonstrate adverse effects from less-than-chronic durations of exposure. Based on chemicals reviewed by MDH for the 2009 HRL rule revision, none had less-than-chronic RfDs that were more stringent than the chronic; however, there exists a need to address all potential effects and more accurately evaluate life stages with higher exposures. It is the use of the higher less-than-chronic exposure rates for drinking water that can drive the final risk values for these durations to be more stringent than for chronic durations (MDH 2008a). The details of this application are described in Section V.B.c.

Supplemental Algorithm for Developmental Susceptibility.

c. Cancer evaluations

Cancer evaluation has also improved considerably since HH-WQSs were first adopted. As described in this section, advancements have been made in identifying more steps and mechanisms that can lead to cancer and enhanced studies to identify pollutants that may act through these mechanisms. An evaluation process once built on broad classifications of a pollutant's carcinogenic potential (Class A to E), now more comprehensively describes the available data, provides weight-of-evidence for cancer potential, and identifies data gaps. Also as thoroughly reviewed by MDH (2008a), closer scrutiny of the available data on life stage sensitivities have shown that exposure to carcinogens during infancy and childhood carries greater risk than for adults and has resulted in the specific approaches to reduce that risk (e.g., ADAFs and higher intake rates).

Technical background

EPA defines cancer as: a disease of heritable, somatic mutations affecting cell growth and differentiation and characterized by an abnormal, uncontrolled growth of cells (USEPA 2006b). In this definition two key characteristics distinguish cancer processes from noncancer:

1) Uncontrolled cell multiplication resulting from alterations to genetic or other cellular targets that control replication or programmed cell death; and
2) The mutations need to be passed on to the daughter cells or are "heritable."

Carcinogenesis as described in EPA's Integrated Risk Information System (IRIS) results from an "event that modified the genome and/or other molecular control mechanisms", followed by "production of a benign or malignant tumor" (USEPA 2010b). Tumors or neoplasms (current terminology) occur as the number of abnormal cells multiplies. Malignant neoplasm means that the abnormal cells are found outside the original tissue site. In the past, the presence of neoplasms in laboratory animal cancer assays was the key measure for identifying carcinogens; cancer assays and human population studies now examine a range of evidence regarding a pollutant's carcinogenic potential from genetic mutations in DNA to specific steps in cell control processes to neoplasm types and tissues. In addition, more understanding on mechanisms has strengthened both identification of carcinogens and improved methods in risk assessments.

Linear carcinogens

Most carcinogens are considered linear carcinogens (response varies directly with the dose), and therefore follow the conservative assumption that no exposure to a carcinogen is without risk. Because cancer can develop from a single mutation in DNA, it is possible that any exposure to a mutagenic carcinogen could result in cancer. However, there are a large number of chemicals that are not known to be mutagens, but have uncertain mechanisms of action leading to cancer through other nonmutagenic modes of action (MOA). Therefore, in the absence of information showing that there are low doses or a threshold at which there is no cancer risk, these chemicals are also assumed to be linear.

The non-threshold, linear assumption in cancer risk assessment uses a cancer slope factor (CSF, same as cancer potency factor, q1*) to calculate risks. The CSF, or cancer incidence per unit dose, is derived from determination of a point of departure (POD), or the lowest estimated dose-response point using modeled toxicity data shown to induce neoplasms or precursor effects; the POD is then used to extrapolate responses at lower doses down to zero. A CSF is an upper bound value for the number of cases of cancer estimated from a lifetime of exposure to a chemical. Therefore, the risk for carcinogens is expressed as a probability of developing cancer, measured as an incremental cancer risk.

For Minnesota's environmental health protective standards, an incremental or additional lifetime cancer risk level greater than 1 in 100,000 (1/100,000 or 10^{-5}) is generally viewed as elevated. This same cancer risk level is used by MDH and many states in EPA Region V. It is the risk level adopted by the Great Lakes Initiative for all the Great Lake States (USEPA 1995b). In general EPA's Office of Water allows states to use cancer risk levels from 1 in 1,000,000 (10^{-6}) to 1 in 100,000 (10^{-5}); however, with assurance that no defined subpopulation would experience cancer risk greater than 1 in 10,000 (USEPA 2000b). The use of the conservative incremental risk levels provides a reasonable approach to developing HH-WQSs given that level is developed based on only exposure from surface water uses that doesn't reflect additional cancer risk from other routes or sources of environmental exposure to that pollutant. Other cancer risk levels calculated for other routes or sources of exposure would be considered additive to the surface water incremental risk; so in total, additional cancer risk for a single pollutant would remain relatively low if that pollutant and other environmental carcinogens are regulated at least to this same risk level. This incremental risk level is also considered against the much higher lifetime background cancer rate (1 in 2 or 50,000 in 100,000) (MDH 2008a). HH-WQS with carcinogenicity as an endpoint will continue to be based on an *incremental cancer risk* ceiling of 1 in 100,000 in recognition of these reasons and current and past policy implementation of this value in environmental standards regulations.

Nonlinear carcinogens

Alternately, MDH or EPA may determine based on sufficient MOA data that some carcinogens exhibit a threshold, or in other words, are nonlinear. For these chemicals cancer risk is not directly proportional to dose levels and there is a level of exposure below which there is no cancer risk. Instead of using the POD to calculate a CSF, as for linear carcinogens, the POD is used to calculate a RfD. Derivation of the RfD for a nonlinear carcinogen (NLC) is consistent with derivation of any other RfD. The final RfD then will be below the level of concern for cancer and may be based on an effect that may be a precursor to cancer or a more stringent threshold for

other noncancer adverse effects. The RfD is then applied in the noncancer algorithm. The nonlinear approach is only used if the MOA is known, the chemical exhibits a threshold, and does not demonstrate mutagenicity (USEPA 2005a). Carcinogenic pollutants with HH-WQS adopted in Minn. R. chs. 7050 and 7052 will now be more precisely identified as linear carcinogen (C) or nonlinear carcinogen (NLC).

Historical and new cancer classification

Carcinogens are classified with a weight-of-evidence (WOE) descriptor which describes the extent to which data support a substance's potential to cause cancer in humans, independent of its carcinogenic potency (IRIS). In 2005, EPA revised the WOE descriptors for new chemical assessments. Historically, the 1986 *Guidelines for Carcinogenic Risk Assessment* provided the original WOE categories from Group A for known carcinogens through Group E for chemicals with evidence of noncarcinogenicity. A more narrative approach assessing all the individual lines of evidence, including a one to two-page summary that explains human carcinogenic potential, is used along with revised WOE descriptors in the more recent 2005 *Guidelines for Carcinogenic Risk Assessment*. The five descriptors are: "*Carcinogenic to Humans*," "*Likely to Be Carcinogenic to Humans*," "*Suggestive Evidence of Carcinogenic Potential*," "*Inadequate Information to Assess Carcinogenic Potential*," and "*Not Likely to Be Carcinogenic to Humans*." MDH uses EPA-derived slope factors for carcinogens in the Group A and B categories under the 1986 guidelines, and "*Carcinogenic to Humans"* and "*Likely to Be Carcinogenic to Humans"* under the 2005 guidelines.

The 1986 guidelines contained a Group C descriptor – *Possible Human Carcinogen* within the classification. A chemical labeled Group C remains in that classification until it undergoes a reassessment under the new guidelines. MDH had historically handled Group C chemicals by calculating a noncancer HRL using an additional uncertainty factor of ten to address potential carcinogenicity. The 2009 HRL Rule provides for a case-by-case formal evaluation of the evidence of carcinogenicity of Group C chemicals. Evaluations and classifications from other agencies is one of the criteria MDH uses to assess carcinogenicity. The National Toxicology Programs classifies chemicals into two categories: "Known to Be Human Carcinogen" and Reasonably Anticipated to be Human Carcinogen" (National Toxicology Program 2005). The International Agency for Research on Cancer (IARC) classifies chemicals into five groups based on carcinogenic potential (IARC 2006). Based on MDH review committee's recommendation, a separate uncertainty factor may be used when a noncancer HRL is derived for a chemical for which evidence of carcinogenicity is strong, but still insufficient to derive a CSF. MPCA will base their toxicological values for Group C chemicals in future HH-WQS on recommendations from MDH.

Early-life sensitivity

At the same time as the 2005 *Guidelines for Carcinogenic Risk Assessment*, EPA published the *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens*. This document provides specific guidance to address the potential for differential risk of early life-stage exposure by adjusting upward the CSF for carcinogens that act through a mutagenic MOA. Although there is much unknown about children's cancer susceptibility, MDH agrees that the data indicate that for many carcinogens there is evidence of early-life sensitivity, and a lifetime of exposure is not necessary for cancer to develop (MDH 2008a). EPA applies the following age dependent adjustment factors (ADAF) to carcinogens with a mutagenic MOA to account for evidence that cancer risk is higher from early-life exposure than from similar exposure durations after age 16:

- ADAF of 10 for birth to less than 2 years of age
- ADAF of 3 for 2 to less than16 years of age
- ADAF of 1 for greater than 16 years of age (applied up to 70 years of age in HH-WQS).

Unlike EPA, MDH has chosen to apply the adjustments to the CSF for all linear carcinogens in their HRL rule, regardless of the mechanism of action. MDH came to this decision in part based on comments from EPA's Science Advisory Board (USEPA 2004a) and the MDH's external Expert Advisory Panel (MDH 2008a). The Science Advisory Board suggested EPA apply a default approach to both mutagenic and to nonmutagenic chemicals for which the mechanism of action remains unknown or insufficiently characterized (USEPA 2004a). Mutagenicity is not the only mechanism of action that provides evidence of early-life sensitivity. Because of uncertainties surrounding mechanism of action for many chemicals and the significance of early-life sensitivity, MDH has chosen to make a policy decision at this time to apply the ADAFs to all linear carcinogens. However, if there is chemical-specific data available that indicate use of an alternative lifetime or no early-life adjustment factor is appropriate, MDH will adjust the CSF accordingly.

d. Addressing toxicological evaluations for breakdown products

MDH also has determined to be health protective, breakdown chemicals that originate from a "parent" chemical should be assessed the same as the "parent" when toxicological data are insufficient for a chemical-specific health based water value (Minn. R. 4717.7900). For some pollutants, when introduced into the environment they undergo chemical transformation from microbial, photolysis, or other processes. Particularly for pesticides, there are known common environmental breakdown products referred to as degradates. While pesticide degradates may have requirements for toxicity testing, sometimes it can be more limited and not meet MDH's database criteria. Similar chemical structures between parent and degradate also adds credence to this approach, which is used by EPA as toxicity screening tools (e.g., Toxic Substance Control Act program). To address degradates found in water resources, MPCA will also apply parent HH-WQSs to environmental degradates when HH-WQSs or MDH health-based guidance values are unavailable.

C. Exposure evaluation

a. Introduction

Exposure assessment is a detailed process that includes the routes a pollutant is taken in (inhalation, ingestion, dermal, etc.), its intake rate (amount on a body weight basis or estimated external dose), and its fate in an organism (metabolism, accumulation, etc.) or toxicokinetics. The calculation of HH-WQSs requires the use of several exposure assumptions in the absence of pollutant-specific data. Exposure in this context refers to the intake rate of a pollutant estimated to be taken in by a specified population as part of the beneficial uses of surface waters (i.e., source of drinking water or incidental ingestion during recreational activities) and aquatic biota (fish consumption). As described in detail in the following sections, MPCA has examined and in some case revised the exposure parameters adopted in 1990 as part of the

revisions to HH-WQSs, primarily because of better data and newer approaches for ensuring protection for developmental life stages.

b. Drinking water consumption: new MDH age and duration adjusted intake rates (IRs)

Standard risk assessment practice has been, and continues to be for EPA in their water programs, to use a drinking water intake rate of 2 liters per day applied with a 70 kg adult body weight (or on a per body weight per daily basis: 0.029 L/kg-day). MDH's 2008 Health Risk Limit (HRL) rule revision improved and replaced these standard default values by developing drinking water intake rates that directly incorporate body weight, use the latest national survey data, account for life stage exposure differences and durations, and enhance protection through use of higher percentiles of exposure. MDH's research clearly indicates that use of one adult-based water intake rate may not be protective of children, especially for formula-fed infants, whom drink much more per body weight than adults (MDH 2008a). MPCA's revised exposure parameters used to account for drinking water consumption when surface waters are designated as a source of drinking water is the drinking water Intake Rates (DWIR)[25] and will be set at drinking water intake rates (IRs) used by MDH in their HRL rule and program.

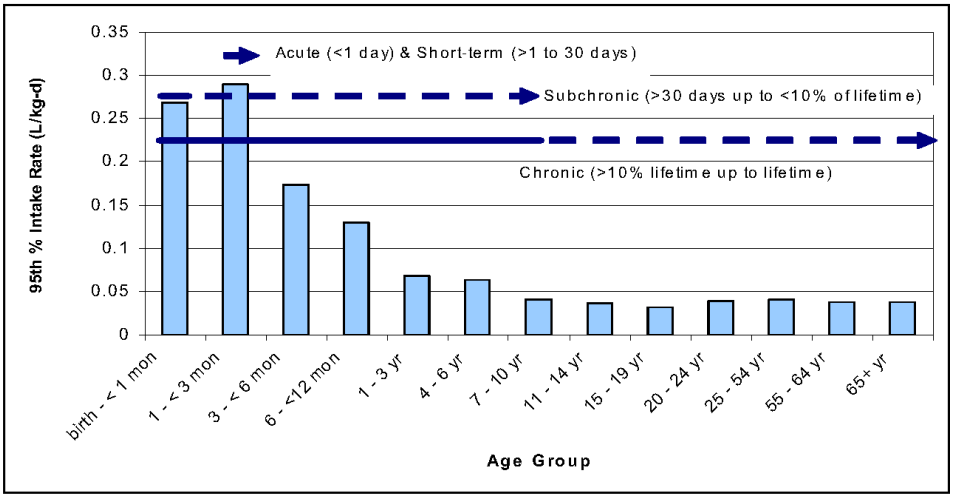
As described fully in MDH's SONAR (2008a), reliable national survey data published by EPA in 2004 on the general population's drinking water sources and uses served as the basis to set new IRs. In addition to adjusting the drinking water rates to be protective of higher intakes by infants and children, the exposure duration-specific intake rates when combined with duration-specific RfDs will also protect against adverse effects that can occur from shorter duration exposures; therefore, instead of only accounting for chronic exposures, the expanded exposure durations (acute, short-term, and subchronic) more appropriately fit with intake rates for the life stages that are most sensitive.

The data used to develop MDH's IRs was the Continuing Survey of Food Intakes by Individuals (CSFII) conducted in 1994 to 1996 and 1998. An important component of the survey is to examine drinking water sources and intake rates for the general population of the United States (USEPA 2008). The data had been analyzed many ways by EPA, including a complete set of statistics from respondents that consumed water on one or both days of the survey, classified as "Consumers Only" results. MDH's rates used "Consumers Only" data by age grouping from birth through adulthood for the durations specified for RfDs to develop four IRs for use with the four duration RfDs and four IRs for application with cancer algorithms (MDH 2008a).

The data tables used came from EPA evaluations of the CSFII (USEPA 2004b, USEPA 2008). For the derivation of noncancer HRLs, MDH selected the following default duration-specific intake rates: acute or short term—0.289 L/kg-day, based on the time-weighted average (TWA) of the 95th percentile intake from birth up to 3 months of age; subchronic—0.077 L/kg-day, based on a TWA of the 95th percentile intake from birth up to 8 years of age; and chronic—0.043 L/kg-day, based on TWA of the 95th percentile intake over a lifetime of approximately 70 years of age (see Figure 2).

[25] The acronym used for HH-WQS is DWIR as these standards incorporate other intake rates in the algorithms (incidental water and fish consumption) as compared to HRLs that are only based on drinking water exposure and use the more generic descriptor IR (Intake Rates).

Figure 2: Duration-specific intake rates (from MDH 2008a)



For the derivation of HRLs for linear carcinogens, MDH adopted EPA's approach for integrating age-dependent sensitivity adjustment factors and exposure information. The default IRs corresponding to the ADAF age groups used in deriving cancer HRLs are based on the TWA of the 95th percentile intake rate for each age range. The values are 0.137 L/kg-day (birth up to 2 years of age), 0.047 L/kg-day (2 to up to 16 years of age), and 0.039 L/kg-day (16 years of age and older). When the ADAFs are not applied, MDH uses the same IR as for the chronic noncancer duration: 0.043 L/kg-day, based on TWA of the 95th percentile intake over a lifetime of approximately 70 years of age.

These default intake rates may not be used if chemical-specific information exists that indicate a different duration or intake rate would be more appropriate. Alternate rates would be based on national survey data to match toxicokinetic *in vivo* data on time for the pollutant to reach steady state and body burden thresholds that elicit an adverse effect. The continued availability of biomonitoring data, coupled with survey and modeling data one exposure patterns, will also improve future exposure parameters. Biomonitoring is the actual measurement of pollutants in human blood, tissues, and excreta.

c. Incidental ingestion of water

HH-WQSs are designed to be protective for surface water uses, including, as previously described, drinking and fish consumption and recreational activities, such as swimming, wading, and water skiing. Water not purposely ingested, but taken in during recreational activities (incidental ingestion) has been a principle component for HH-WQSs for microbiological pollutants (e.g., *E.* coli) that protect from gastrointestinal illness. However, for protection from toxic pollutants incidental ingestion has been considered to be less significant in terms of other routes of greater exposure. EPA and states have differed in their use of incidental ingestion in their development of water quality criteria and standards with some EPA programs (Great Lakes Initiative) and states (including Minnesota) specifically accounting for this exposure route and amount, while others have not; MPCA did adopt incidental ingestion as part of the methods to develop HH-WQSs in 1990 based on general assumptions at that time (Minnesota Pollution Control Agency (MPCA) 1989).

Although exposure from incidental ingestion is likely to be highly variable and difficult to quantify, MPCA feels it is important to recognize this potential source by including it in the algorithms for some surface waters. The incidental ingestion exposure parameter applies for HH-WQSs developed for waters not designated as sources of drinking water, where the beneficial uses are narrowed to fish consumption and recreation. Standards that address drinking water exposure apply a drinking water intake rate (DWIR), and adding to that the smaller incidental ingestion amount wouldn't significantly increase the rate. However, when DWIR is not part of algorithm for standards applied in waters only protected for fish consumption and recreations, for less bioaccumulative chemicals, the incidental ingestion amount becomes significant and important to include as an intake rate. EPA has concurred that incidental ingestion can be important in states like Minnesota with high swimming and boating activities and included this exposure parameter in the AWQC for the Great Lakes States.

The amount used currently in HH-WQS methods is 10 mL/day and was based on the precedence of its use in other EPA Region 5 states and the GLI methods for human health (Minn. R. ch. 7052) and by comparing it to an estimated mouthful (30 to 50 mL) of water when adopted in 1990. This incidental ingestion amount used the 70 kg default body weight, equating to a per body weight incidental ingestion intake rate of 0.00014 L/kg-day. This is the same rate adopted in 1998 in Minn. R. ch. 7052.

The new proposed incidental ingestion intake rate (IWR) of 0.0013 L/kg-day is calculated based on a reasonable maximum estimate of water related activities in Minnesota (based on MDH 2006 and *Risk Assessment Guidelines for Superfund*; RAGS (U.S. Environmental Protection Agency (USEPA) 1986a)) and a swimming ingestion rate from EPA's 2011 *Exposure Factors Handbook* on a per body weight basis (USEPA 2011). Children ages 1 to 17 were assumed to swim twice per day for 0.5 hours for six days a week for three months (12.9 weeks) in summer. This amounts to 77.4 days per year, converted to 77.4 hours per year of swimming (MDH and ATSDR 2006). An upper percentile (97^{th} %) water ingestion rate of 120 mL/hour for swimming activities for children under 16 is used (USEPA 2011). A body weight estimation was calculated from a table of recommended values for different age categories (USEPA 2008) and is an average for ages 1 to < 9 to be protective of young children; and covers the life stages of highest exposure for a defined chronic duration of eight years (>10% of a human lifespan).

$$IWR_{(year)} = \frac{SI \; x \; EF}{BW}$$

Where:

IWR – Incidental water ingestion rate in L/kg-day

SI = swimming ingestion: 120 mL/hour (97^{th} percentile water ingestion rate for exposure scenarios involving swimming activities for children under 16 - USEPA 2011, Table 3-5)

EF = exposure frequency: 77.4 days/year (6 days/week during 12.9 weeks/year – MDH and ATSDR 2006)

BW = body weight: 20.1 kg (average body weight for 1 to <9 years – USEPA 2008, Table 8-22 as provides more age ranges also from NHANES than USEPA 2011)

$$IWR_{chronic} = \frac{\frac{120\ mL}{hour}\ x\ \frac{1\ L}{1{,}000\ mL}\ x\ \frac{1\ hr}{day}\ x\ \frac{77.4\ days}{yr}\ x\ \frac{1\ yr}{365\ days}}{20.1\ kg} = 0.0013\ L/kg - day$$

d. Other routes of exposure considered for recreational use

Introduced in a number of sections related to exposure is the fact that a pollutant's physicochemical characteristics play a significant role in determining the ways humans are going to be exposed to a pollutant in surface water; for example many nonionic hydrophobic organic pollutants partition to lipids, biomagnify in the aquatic food chain, and end up in high concentrations in fish tissue. Volatile Organic Chemicals (VOCs) are known as "volatile", because of their tendency (characterized by the Henry's Law Constant) to readily leave surface water to partition into the air. EPA AWQC and HH-WQSs have historically been based on accounting for two to three routes of exposure for surface water pollutants—all based on ingestion: drinking and incidental water intake and fish consumption; however, it is known that other routes of exposure are possible when a pollutant is in surface waters, specifically dermal and inhalation (USEPA 2000b).

There are a few main reasons why dermal and inhalation exposure estimates are not explicitly included in HH-WQSs: overall HH-WQSs methods protect humans in most scenarios and there is large uncertainty in standardizing exposure assessments and fewer relevant toxicological values for these pathways. HH-WQSs are upper limits for a pollutant in surface water that should not be exceeded to ensure designated beneficial uses are met. In the case of HH-WQSs, the beneficial uses are based on human consumption of drinking water and fish and recreational activity (e.g., swimming, boating, and water skiing). For microbiological criteria, the risk of illness has always been associated with incidental ingestion and representative thresholds of bacterial organism (USEPA 1986b). EPA and the states have approached recreational exposure for toxic pollutants in different ways: through the inclusion of incidental water intake, by not using recreational exposure rates and instead relying on the use of drinking water or fish consumption to be sufficiently protective, or through application of the Relative Source Contribution (RSC) factor (USEPA 2000b). Since 1990 HH-WQSs have included an incidental ingestion rate and general reliance on use of default RSC of 20% to address recreational exposure and its limitation.

As part of the HH-WQSs revisions, the available options for addressing dermal and inhalation pathways was reviewed, but not included in the revised algorithms, because broad or default application is complicated by many factors. As described in EPA's Superfund Risk Assessment Guidance (RAGs, Part E) (USEPA 2004c), dermal evaluation is based on limited pollutant-specific data on active and passive mechanisms of absorption, on metabolism processes in skin, and toxicological values from dermal administration studies. EPA's RAGs have screening level methods to assess dermal exposure from water and soil, but these have not been adopted for use as the basis for EPA AWQC. Developing an approach to address the inhalation pathway for swimming in surface waters is even more complicated. The swimming and wading scenarios could include inhalation exposure for some pollutants, particularly for VOCs; however, standardizing an approach is influenced by many environmental factors (e.g., wind speed and

air and water temperature patterns) and wide variation in swimming and wading activity patterns (USEPA 2008). EPA RAGs and Office of Pesticide Programs and other federal programs (ATSDR) have screening methods to make reasonable estimates of risk from all exposure pathways when swimming or recreating in pools or surface waters (USEPA 1989, USEPA 2003c, MDH and ATSDR 2006); while, no approach was deemed complete or reliable enough for statewide application into rule, MPCA and MDH have completed site-specific evaluations of recreational use when warranted to supplement HH-WQSs (MDH and ATSDR 2006; MDH 2008b; MPCA 2008).

MPCA recognizes that the consideration of dermal and inhalation can be significant contributors to swimming exposure for certain classes of chemicals (e.g., VOCs and inhalation; PAHs and dermal) if *actually* present in surface water or sediments; for this reason, MPCA has requested and will continue to request MDH *Health Consultations* on a site-specific basis to evaluate recreational use protection with known contamination at public beaches and other high swimming use areas as described in Section V.C.b. (MDH and ATSDR 2006, MDH 2008a). To address these routes in HH-WQS methods, MPCA has updated and increased their IWR to better reflect reasonable swimming and wading scenarios in Minnesota and use newer exposure data for children (see the previous section, *Incidental Ingestion of Water*). As part of the noncancer evaluation and definition of RSC, MPCA will apply and set this value to address dermal and inhalation routes when relevant to a pollutant through the application of the lowest default RSC value of 0.2 (Section IV.C.h.). MPCA will continue to examine this area of risk assessment to determine in later revisions if other approaches are available. In addition, as described in the *Protection Level Goals*, reliance on these approaches for setting parameters for HH-WQSs helps to address uncertainty and ensure protection for all beneficial uses (see Sections VI).

e. Fish consumption rates

Introduction

An important component of developing methods for human health protection from pollutants in surface water and fish in 1990 was the choice of an appropriate and protective fish consumption rate (FCR). EPA protocol at that time had focused on an average level of consumption for the general population (consumers and nonconsumers of fish and shellfish) of 6.5 grams of fish consumed per person per day (g/d) (USEPA 2000b). MPCA, with the assistance of a Toxics Technical Advisory Committee (Appendices B1 and B4), determined that in Minnesota the importance and popularity of fishing warranted a higher level of protection and use of regional data on fishing habits. By focusing on regional, freshwater fish consumption surveys of adults, a FCR of 30 g/d was developed based on approximately the 80th percentile rate (80% of the population surveyed ate 30 g/d or less of freshwater fish) from this population of fish consumers, which equated to about a 90th percentile rate for the general population at that time.

Since MPCA adopted the human health methods in 1990, additional survey data have been published on regional and national FCRs. Because EPA supports the efforts of states and tribes to maintain currency in their HH-WQS, many of the studies with relevance to Minnesota's FCR

have been evaluated by EPA. As described in detail in this section, MPCA has completed a limited reviewed of the ongoing use and application of the existing 30 g/d based on adults in light of the more recent data provided in these EPA evaluations. The reviewed centered on EPA published evaluations, primarily ,their recent republication of results from a fish consumption survey conducted in Minnesota and additional studies and guidance on application of national survey data for state and tribal risk assessments (Tran et al. 2013, USEPA 2013a, USEPA 2013b). The review focused on the relevance of this more current EPA information on the ongoing use of the 30 g/d as the default FCR.

However, to meet the key objectives of the method revisions to develop improvements for protecting infants, children, and other developmental life stages, a comprehensive review was conducted on available survey data to examine fish consumption patterns and rates for children ages 1 to 18. The review examined fish consumption differences in children and adults and short-term rates. Identification of adverse effects that can occur after less-than-chronic durations of exposure, which form the basis for multiduration RfDs and new ADAF windows of higher cancer susceptibility, has important implications for FCRs used in HH-WQSs. MPCA has conducted this review by considering MDH's approach for developing new drinking water intake rates for infants and children based on less-than-chronic durations and examination of the timing of highest exposure (per body weight), within the context of differences in fish consumption profiles across developmental life stages.

The main elements consider for both of the limited review of information relevant to MPCA existing FCR based on adults and new comprehensive evaluation of children's FCRs are listed below, with longer summaries following:

- *Use of Regional or Minnesota-specific Survey Data*: Consistency with MPCA's previous approach of using regional to local fish consumption data;
- *Higher Level of Protection than National Defaults*: Continuation of having a higher level of protection based on people that catch and eat fish (referred to as "anglers" or "fishers" in EPA 2000b or "consumers" in other EPA guidance);
- *Examination of More Recent Survey Data*: Review of more recent fish consumption data published by EPA with relevance to Minnesota's WQSs;
- *Use of Children's Survey Data*: Examination of the need for shorter duration and potentially higher fish consumption rates to reflect children living in fishing households that may eat more fish than average and may have higher exposure when considered on a per body weight basis; and
- *Revised Fish Consumption Rates*: Proposed application of the existing adult FCR, newly derived children's FCRs, and approach for developing alternate FCRs if needed on a site- or pollutant-specific basis.

Regional Fish Consumption Survey Data: Minnesota's existing adult, sport fish consumption rate

Fishing on Minnesota's lakes, including Lake Superior, and rivers is (and long has been) very popular; therefore, the fish consumption habits of the fishing or angling population has been the basis for determining the original fish consumption rate for use in HH-WQS (see

details in Appendix B1). Survey data specifically collected on Minnesota anglers was not available when the existing FCR was developed. However, survey results had been recently published on the amount of fish eaten by anglers in Wisconsin and Ontario and were used to reflect expected similarities in fishing and eating patterns in Minnesota freshwater, caught fish consumers (see Appendix B). The amount of freshwater fish consumed by anglers varies from none to more than one meal every day. The 80th percentile FCR was selected as the value to calculate HH-WQSs in 1990; the mean of the 80th percentile values from the two surveys was about 30 g/d[26]. A later and larger regional, fish consumption survey of anglers in Michigan also supported a value of close to 30 g/d as representative of the 80th percentile FCR for this population of sport, fish consumers (West et al. 1989, USEPA 1997, USEPA 2011). The Michigan survey data were used to derive the fish consumption amount used by the Great Lakes Initiative (GLI) of 15 g/d, which was the average consumption rate from that survey.

Thirty grams per day is also equivalent to approximately one half-pound meal of freshwater fish per week, or 26 pounds a year. MDH bases Fish Consumption Advice[27] for adults on a one half-pound meal (or 270 g) per week for a 70 kg person.

Higher Level of Protection: Minnesota's greater protection goals for fish consumers

As summarized, Minnesota's fish consumption rate of 30 g/d represented the 80th percentile rate from two regional surveys of freshwater, sport fish intake. When compared to freshwater FCRs estimated for the general population (fish consumers and nonconsumers) at that time, it was closer to a 90th percentile rate (MPCA 1989).

Since adoption of the 30 g/d, EPA has provided additional guidance to consider in developing and revising HH-WQSs. In EPA's 2000 guidance, a new national fish consumption rate of 17.5 g/d was developed to replace EPA's previous national default rate of 6.5 g/d (USEPA 2000b). EPA's stated protection goals focus on the "high-end exposures for the general population." Their new rate was the estimated 90th percentile *per capita*[28] or general population fish consumption rate for freshwater and estuarine fish and shellfish. The source of the fish consumption data was the U. S. Department of Agriculture (USDA) CSFII; the latest surveys were conducted from 1994 to 1996, with a special survey completed in 1998 specifically to obtain more data on children.

EPA's 2000 guidance also provided recommendations for considering subpopulations of fish consumers: women of child bearing age (15 to 44 years of age) and subsistence fishers. EPA again used the data available from the CSFII surveys, but used higher percentiles from both the "consumers only" and "*per capita*" results as representative FCRs based on the 90th percentile (165.5 g/day) and the 99th percentile (142.4 g/d), respectively (USEPA 2000b).

[26] FCR is also used in the context of per body weight. Both the Minnesota FCR of 30 g/d and GLI FCR of 15 g/d were applied with an average adult body weight of 70 kg (150 lbs.).
[27] For more details see, http://www.health.state.mn.us/divs/eh/fish/index.html .
[28] *Per capita* in this context refers to the fact that the data are based on fish consumption rates for all adults surveyed whether they eat fish or not (EPA 2002).

The rate provided for subsistence fishers was also characterized as an average rate of fish consumption from a range of actual surveys of tribal fish consumers (including studies of tribal harvest of freshwater, estuarine, and marine fish and shellfish).

Recently, EPA has provided additional information on their recommended protection goals (USEPA 2013a); however, not included was specific rationale or discussion of the appropriate use of survey data to address regional fish consumption differences, survey limitations, or importance of pollutant-specific toxicological profiles when setting meaningful upper-percentile and appropriate FCRs. Use of specific percentiles for intake rates are dependent on the available datasets and state-specific policies for supporting the goals of protective environmental standards (USEPA 2000b). In accordance with State policy and EPA guidance, MPCA continues to strive to use the best available survey data to set FCRs representative of upper-end freshwater, caught fish consumers within the context of their specific application in HH-WQS algorithms (less-than-chronic versus chronic; statewide versus site-specific application; and pollutant-specific toxicological considerations). All these considerations in regards to better application of the existing adult FCR, development of new children's FCRs, and approach for future alternate FCRs are fully described in the following sections.

Examination of More Recent Survey Data: 1990 to 2013 Overview of fish consumption data published by EPA for adults

Since MPCA adopted a fish consumption rate in 1990, EPA has published four reviews pertinent to Minnesota's WQSs:

- *Great Lakes Water Quality Initiative Technical Support Document for Human Health Criteria and Values* (1995b)
- *Exposure Factors Handbook: 2011 Edition and Related Studies: Fish Consumption in Connecticut, Florida, Minnesota, and North Dakota (2013b), and*
- *Methodology for Deriving Ambient Water Quality Criteria for Human Health* (2000b)

While EPA guidance has evaluated many different populations and fish consumption patterns, the surveys and results most applicable to Minnesota's FCRs are based on consumption of freshwater, caught fish[29]. In the context of "caught", HH-WQS are numeric standards that protect consumers of fish caught from Minnesota's lakes, including Lake Superior, and streams. Because fish caught in Minnesota's waters can be eaten by the person catching the fish, as well as, by members of their household or by friends and community members, the best survey data would include total freshwater fish consumption patterns and rates. Intake of estuarine and marine store-bought fish and shellfish is

[29] Minnesota's HH-WQSs have focused on fish and not other aquatic species for a number of reasons. Based on angler and creel surveys and state fishing regulations, consumption of other aquatic organisms is restricted and generally uncommon. Minnesota's Department of Natural Resources (DNR) fishing regulations do not allow for harvest of any freshwater mussels and other organisms such as crayfish, frogs and turtles tend to be collected for purposes other than consumption (e.g., bait and turtle races) (*Minnesota Fishing Regulations 2011).* Data on other aquatic organisms would still be considered on a pollutant-specific basis if deemed relevant in the context of developing BAFs to protect human consumer (see Section IV.C.g.).

considered in the context of other sources of exposure (specifically under the RSC for noncancer evaluations).

Great Lakes Water Quality Initiative Technical Support Document for Human Health Criteria and Values

Of the relevant EPA guidance documents, regional freshwater, caught fish consumption data were specifically reviewed as part of the GLI criteria (USEPA 1995c). As previously described, the GLI human health criteria are based on a regional mean FCR of 15 g/d. The key survey data came from freshwater, sport fishing surveys in Michigan conducted by West (1989). When MPCA adopted the GLI criteria for Lake Superior, the 30 g/d was adopted in place of the default rate for the same reasons it was adopted statewide, higher protection for Minnesota's fishing population.

Exposure Factors Handbook: 2011 Edition and Related Studies: Fish Consumption in Connecticut, Florida, Minnesota, and North Dakota

While focusing on national-scale, recommended general population total fish and shellfish consumption rates and patterns, the current and past *Exposure Factors Handbooks* (USEPA 1997, USEPA 2008, USEPA 2011) have provided reviews of the most relevant and robust survey data available to develop freshwater, caught FCRs. EPA has recognized the surveys from Wisconsin and Ontario that were the basis for Minnesota's current FCR and the Michigan surveys by West et.al. from 1989 and 1993 as key surveys for estimating FCR for recreational fishing populations. The most recent survey identified by EPA as relevant and included Minnesotans who report eating self-caught fish was the 2001 Benson, et al., survey (described relevant to adult data in Moya 2004, USEPA 2011, Sections 10.3.2.7. and 10.5.14 under Westat 2006 and Moya et al., 2008, and USEPA 2013b).

The 2001 Benson, et al. survey was conducted in Minnesota and North Dakota in October and November 2000 (Moya et al. 2008, USEPA 2013b). The survey targeted residents with fishing licenses, members of Tribal communities, and new mothers, as well as, the general population. This survey has advantages over the national survey data, because the people surveyed specifically included Minnesota freshwater fish consumers. However, because the survey had a very low response rate (15 to 20%) and lack of sufficient follow-up on biases for nonrespondents, the results of this survey were not considered acceptable for directly developing updated FCRs for HH-WQS (USEPA 1998, Maschwitz and Preimesberger 2009). These two limitations, in particular, affect the reliability of using the respondents' results to extrapolate FCRs to the broader populations of Minnesota fish consumers they were sampled from. However, MPCA considered this data as important for providing context to examining the existing adult FCR and children's fish consumption rates (see discussion, *Use of Children's Survey Data*).

EPA has completed a number of reassessments of the 2001 Benson, et al. survey for understanding patterns and rates of fish consumption by regional subpopulations and how they are similar to or different from national general population survey results (Moya 2004, Moya et al. 2008, USEPA 2011, USEPA 2013b). EPA has presented information on mean and upper percentiles of FCRs and fish species consumed by a number of factors (e.g., age,

ethnicity, gender, and income). The most relevant presentation of the survey data are in the context of freshwater, caught rates. EPA had determined the rates for the average and 95th percentile from all the consumers surveyed, including children, in Minnesota were 14 g/d and 37 g/d, respectively (USEPA 2011). Where EPA provided results separately for children and adults, the 95th percentiles were 13.7 g/d for children from birth to age 14, 39.8 g/d for males ages 14 and over, 24.9 g/d for females ages 15 to 44, and 37.2 g/d for females ages 44 and over (USEPA, 2011, Table 10-84).

Because intake rates are more accurate when considered on a per body weight basis by respondent, the survey data are considered further in units of grams of fish consumed per kilogram of body-weight-day (g/kg-d). For comparison, the existing adult FCR of 30 g/d with a 70 kg body weight equates to 0.43 g/kg-d. The statistics EPA presented from the Benson et al. 2001 survey did not include rates for children separate from adults for freshwater, caught fish consumption. Rates were provided for children and adult survey respondents combined, but as there are significant differences in the rates for younger age groups from those for older age groups, as further described in *Use of Children's Survey Data*, those rates were not appropriate for examining adult only information. However, an estimate could be made for adults in the subgroups surveyed using the statistics provided on all fish consumed (uncooked weights) for adults and children in Table E-178 in USEPA 2013b. The average percent of freshwater, caught fish consumed from all fish of 42% was determined using the results presented in Table 10-84 in USEPA 2011.

The 90th and 95th percentile estimates for the subpopulations surveyed in g/kg-d, respectively, were 0.31 and 0.53 for general population (consumers and nonconsumers), 0.35 and 0.53 for anglers, 0.28 and 0.41 for new mothers and other adults in their household, and 0.23 and 0.32 for Native Americans from the Bois Forte Tribe. Based on adjusting the 90th and 95th percentile adult rates from Table E-178 to estimate freshwater, caught FCRs, the existing adult rate of 0.43 g/kg-d was in this range for all subpopulations surveyed. These percentiles also agree with those for sport-caught fish rates shown in USEPA 2011 Table 10-84 in g/d when compared with 30 g/d.

Besides the Benson et al. 2001 survey, the most recent EPA *Exposure Factors Handbook* also summarized other studies with relevance to freshwater recreational and Native American fish consumers (USEPA 2011, Sections 10.5 and 10.6 and Tables 10-5 and 10-6). Of the available survey data evaluated by EPA, no single survey or group of surveys provided recent and reliable enough information for EPA to develop recommended FCRs for freshwater, caught consumers by state, region, or subgroup. However, the survey information from states and tribes with similar geographical and freshwater only fishing resources was reviewed for providing some context to Minnesota's existing adult FCR.

Of the surveys summarized, those with potential relevance to Minnesota's recreational fishing population included Wisconsin (Fiore et al. 1989 – this study was already used by MPCA when originally developing the 30 g/d), Michigan (West et al. 1989 and 1993), New York (Connelly et al. 1996), Indiana (Williams et al. 1999 and 2000), North Dakota (Benson et al. 2001) and Tennessee (Campbell et al. 2002). Native American populations surveyed in Wisconsin (Peterson et al. 1994), New York (Fitzgerald et al. 1995 and Forti et al. 1995), and

North Dakota (Benson et al. 2001) might also provide some information relative to fish consumption patterns in Minnesota tribes. The use of information provided by these surveys, however, is very limited primarily due to the fact that all of these studies, except for Benson et al. 2001 and Campbell et al. 2002, were conducted in the 1980s or 1990s. These older surveys have greater uncertainty for determining the current rates of fish consumers, especially for subpopulations such as women of child-bearing age, anglers, and subsistence fish consumers, given the extensive outreach on both the benefits of fish consumption, as well as, the risks associated with mercury and other fish pollutants. In addition, the most reliable surveys are those based on random samples with high response rates; only a few surveys were random and all had low response rates of 15 to 60% with no or only minimal follow-up on nonrespondents. The surveys also differed in the length of time consumption patterns were examined (short-term versus annually) and in determination of meal sizes (questioned or assumed at 227 g).

Considering the limitations of the surveys reviewed by EPA on freshwater, caught fish consumers (USEPA 2011), there was general support across most studies that the existing adult FCR of 30 g/d is representative of an upper percentile rate. However, two of the four surveys of Native American rates suggested 30 g/d being closer to an average rate. Peterson et al. 1994 of Chippewa Tribal members in Wisconsin and Forti et al. 1995 of Mohawk Nation at Akwesasne in New York determined the average rate of respondents was 39 g/d and 29 g/d, respectively, based on assumed meal sizes of 227 g. This information will be important to consider in future reviews of additional survey data and for alternate FCRs.

Methodology for Deriving Ambient Water Quality Criteria for Human Health

EPA's recommended national default fish consumption rates for use in the 2000 AWQC for human health methods was based on the USDA's CSFII surveys as analyzed in EPA 2002 (USEPA 2000b). The CSFII historically was the key source of national food consumption patterns (USEPA 1997, USEPA 2002a). The most recent fish consumption CSFII data at that time was from 1994 through 1996 and 1998, where a special survey was conducted to obtain more data on infants and children. EPA's recommended national default rate for use by states and tribes when developing HH-WQS of 17.5 g/d was the 90th percentile *per capita* (or "U.S. Population" as labeled in USEPA 2002a) value based on uncooked, freshwater and estuarine fish and shellfish consumption estimates (Section 5.1.1.1 Table 4 in EPA 2002a).

Per capita data are based on all survey respondents whether they consumed fish or not. On a per body weight basis, the comparable 90th percentile, U.S. Population estimate is 0.236 g/kg-d (Section 5.1.1.2, Table 4 in EPA 2002a).

EPA's 2000 guidance for human health also provided recommended default fish consumption rates to use for adult subpopulations: sport fishers, subsistence fishers, and women of childbearing age (15-44). EPA's evaluation of the survey did not specifically include evaluating freshwater, recreational or sport-caught fish consumption, but instead referenced the U.S. general population rate of 17.5 g/d as an average recommended rate for this group. The recommended rate for subsistence fishers was 142.4 g/d based on an upper percentile rate from the national survey, representing an average rate from the range of available studies on Native American Tribes (USEPA 2013a). Default rates for women of

childbearing age for use if a pollutant's toxicological profile showed that *in utero* developmental effects were the most sensitive endpoints were based on the 90th percentile of "Consumers Only" data from the CSFII of 165.5 g/d); the updated intake rate is 172.9 g/d (updated in Section 5.2.1.1. Table 1 in EPA 2002a) and is 2.92 g/kg-d on a per body weight basis (Section 5.2.1.2. Table 1 EPA 2002a)[30].

Another, comprehensive and current source of national data that includes assessment of environmental pollutant exposures and diet is the National Health and Nutrition Examination Survey (NHANES)[31]. The USDA states that information in this survey, *What We Eat In America*, has replaced the CSFII[32]. The latest EPA *Exposure Factors Handbook* (2011) for the first time used the NHANES data from 2003 through 2006 to develop the national general population fish intake rates (USEPA 2011, Section 10.2.1. and Table 10-1). The fish consumption data collected differ somewhat from that collected as part of the CSFII. EPA analysis does not include the source of the fish: freshwater, estuarine, or marine. The results were grouped by total finfish, total shellfish, or both. EPA is currently evaluating the utility of the NHANES datasets for use in developing state or regional freshwater, caught FCR for general population and subpopulations. The results of this analysis will help set the foundation for future updates of FCRs by identifying available and reliable data or confirming gaps in the use of this comprehensive survey for the particular needs of HH-WQS.

EPA guidance used CSFII survey data in different ways to provide upper percentile FCRs in the 2000 guidance, but the appropriate use of these rates in HH-WQS based on the strengths and limitations of the survey data were not fully explained and a more recent evaluation of the national survey data provided different and lower rates when appropriately adjusted from two-day survey statistics as collected in the CSFII and NHANES, more representative of meal rates, to longer or *usual* consumption rates (defined as 30-day averages) appropriate for use in risk assessments (Tran et al. 2013). This newer method for estimating usual intake rates resulted in an average reduction of 15% for means and approximately a 50% reduction in 95th percentile rates (Moya 2013)[33]. Based on data available from EPA from the CSFII and NHANES provided to date, the default FCRs recommended by EPA in the 2000 guidance are not relevant for direct application into HH-WQS given their basis in meal rates, broad application of national survey data for freshwater and estuarine finfish and shellfish data for freshwater, caught rates, and lack of specific guidance on application of national survey data to meet specific protection objectives for freshwater, caught fish consumers. However, the data provided by national surveys can inform FCRs for HH-WQs.

[30] The details on the choice of percentiles, default values, and intended use are not provided in USEPA 2000b. EPA states that the details will be published in an "Exposure Technical Support Document"; however, as of December 2013 that document has not been published.

[31] http://www.cdc.gov/nchs/nhanes/about_nhanes.htm

[32] http://www.ars.usda.gov/Services/docs.htm?docid=13793

[33] Personal communication, Dr. Jacqueline Moya, USEPA, December 3, 2013

Conclusions based on EPA Guidance and Reviews of Survey Data

EPA *Exposure Factors Handbook* (2011) reviewed sources of available survey data for recreational or caught freshwater fish consumption, the population of fish consumers of most interest for HH-WQS. The available surveys summarized in Tables 10-5 and 10-6 provide some information for considering adult and subpopulation FCRs, but differences in survey methods, populations surveyed, older age of studies, and other data limitations, affect their reliability for directly developing updates to Minnesota's adult FCR.

MPCA is also not currently proposing to change the adult FCR for HH-WQSs based on the rates recommended in EPA 2000 guidance (USEPA 2000b). As previously discussed, EPA has not provided complete guidance on how the default rates in the 2000 guidance should be applied based on the basis of the national survey data in short-term meal rates and use to estimate long-term average consumption rates and lack of specific evaluation of freshwater, caught fish statistics within the broader types and sources of fish consumed.

The lack of current and robust survey data on freshwater, caught fish consumption rates and patterns limit the options for refining or confirming the existing adult FCR. However, information available from regionally conducted and national surveys have provided evidence to support the continued use of 30 g/d, better applied as 0.43 g/kg-d, as being representative of an upper percentile FCR for most consumers. The available survey data have also proven useful for considering fish consumption patterns and relative differences between subpopulations of fish consumers, including differences between age groups, as described in the next section on development of a new children's FCR.

MPCA continues to work with EPA and seek public input to determine and review relevant survey data for future additions or refinements of the fish consumption rates for adults and subpopulations.

Use of Children's Survey Data: Development of a children's fish consumption rate

Described earlier in this document under *Defining Developmental Protection – A Key Revision Guiding Principle*, a key objective of the revisions is to update human health-based methods to maintain consistency with MDH's HRL Rule and explicitly account for exposure and toxicological differences identified for the developmental life stages. As discussed already, DWIRs used in the revised HH-WQS algorithms will better match a pollutant's toxicological profile and may be based on shorter durations (less-than-chronic) and adjustments by age. The need and application of developmental protection as it relates to exposure to pollutants from fish consumption requires a different approach as discussed in this section in order to develop alternate FCRs.

The first step taken to examine the need for an alternate developmental life stage FCR at ages when children start to eat fish involved determining what data are available on children's fish consumption patterns, is there a difference between a child's intake and an adult's intake (particularly on a body weight basis), and if so, is there a FCR that can be recommended for use in HH-WQS. MPCA Research Scientists, Dr. David E. Maschwitz and Angela L. H. Preimesberger, conducted the review, which was published in MPCA's *Environmental Bulletin* (2009).

In short, their review incorporated Minnesota and regional survey data on fish consumption by children in fishing households and CSFII (1994-1996, 1998) national survey results more broadly applicable to general population fish consumption patterns (CSFII did not identify sources of fish as sport-caught versus store-bought) (USEPA 2002a, Moya 2004). Basing the review on the same data hierarchy and protection-level goals used to develop the adult FCR, Maschwitz and Preimesberger concluded (2009):

- While there are Minnesota and regional survey data of fishing households that included children, there were limitations to all the surveys that precluded developing a children's FCR using that data in the same manner as was done for the adult FCR.
- However, the regional surveys provided information relevant to assist in defining the children's age ranges that may have higher intake rates than adults and enough data to approximate a range of 7 to 9 g/d (0.43 to 0.59 g/kg-day on a per body weight basis) as being around an 80th percentile fish consumption rate for children ages one to five surveyed.
- The CSFII survey results, because of the large population of adults and children surveyed on fish consumption, provided the best data to examine children's rates on a per body weight basis and by fine age ranges (as referenced previously in the section, *Defining Developmental Protection*) and to develop an alternate approach for setting a children's FCR;
- Because there is a relationship between fish consumption patterns of parents and children in fishing households and the assumption that children will eat about the same serving size of fish independent of the source and type of fish, a comparative approach *(on a per body weight basis)* was deemed the best approach and provided policy basis to develop a children's FCR for use in HH-WQSs.
- Across surveys and the full complement of fish consumption data, "child to adult" consumption ratios on a per body weight basis were approximately two for children ages one to five, with older children having rates similar to those for adults. (Table 4 *in* Maschwitz and Preimesberger 2009).
- Application of a factor of two to the adult FCR of 0.43 g/kg-d provides the basis for a recommended FCR for children ages one to five (ending at age 6) of 0.86 g/kg-d.

In conclusion, the fish consumption surveys did not provide robust regional data sufficient to derive a children's FCR by the same methods used to develop the existing adult FCR; however, the full complement of survey data were sufficient for MPCA to develop a new children's fish consumption rate using a different approach for ages 1 through 5 of 0.86 g/kg-d. This children's FCR reflects an advancement of MPCA policy on the protection of more vulnerable life stages, based on best available fish consumption data (Maschwitz and Preimesberger 2009).

This rate and use of the 1 through 5-year-old age group is also consistent with newer survey data. Besides the comparison in fish intake rates for different ages groups provided in Maschwitz and Preimesberger (2009), the latest EPA *Exposures Factor Handbook* (2011)

using NHANES data provide another dataset for examining fish consumption patterns for children. The available survey results show that there are a small percentage of babies from birth up to 1 year that consumed some finfish (2.6%). More importantly the percentage of finfish consumers increases to 14% in 1 year olds and stayed at that percentage until another increase in participants' consumption at 21 years of age and older (23 to 29%). Also of note from the "Consumers Only" total finfish statistics is that in terms of grams of fish consumed per kg of body weight-day (g/kg-d), the mean intake rates for children in the 1 to < 2, 2 to <3, and 3 to <6 was twice that of all older age groups beginning at age 11 (USEPA 2011, Table 10-1). This is consistent with the difference noted in the CSFII results and other studies that included children and adult rates (Maschwitz and Preimesberger 2009). It is reasonable to consider this data as relevant to freshwater fish consumption as the type of fish eaten, whether store-bought marine or freshwater self-caught, etc., is not likely to significantly change the usual amount eaten per meal (i.e., if a child eats about 2.3 oz. of tuna per meal, it is reasonable to assume that when eating freshwater fish, he/she would also eat about 2.3 oz. of perch, walleye, or other species of freshwater fish).

Compared another way by examining fish meals, using the CSFII survey data on average fish meal sizes for children age 2 through 5 years old of 64 g (2.3 oz.) (Smiciklas-Wright et al. 2002), with an age-adjusted body weight of 17.4 kg, this equates to approximately seven meals per month for this age group. The adult rate of 30 g/day with 70 kg body weight equates to about four-8 oz. fish meals per month. Described below are the considerations of these and other policy and scientific factors used to determine how this rate is used in the revised HH-WQSs.

Revised and New Fish Consumption Rates:

MPCA has reviewed newer information to develop additional FCRs and approaches to strengthen the HH-WQS that recognize both the State goals and EPA guidance, but further considers the basis of the fish consumption data and appropriate application in less-than-chronic versus chronic algorithms and in statewide versus site-specific scenarios. In addition, as new or revised HH-WQSs are developed for pollutants of concern in fish tissue, other FCRs may be more appropriate for application given the duration of the adverse health effects and impacts on developmental life stages. The proposed application of the

existing adult and newly recommended children's fish consumption rates are described in the context of noncancer and cancer algorithms. The FCRs and their applications as described below are relevant for use in development of both water and fish tissue HH-WQSs.

Adult Fish Consumption Rate used for Chronic Noncancer and Cancer Algorithms

MPCA is not proposing to change the existing adult FCR in Minn. R. chs. 7050 and 7052. MPCA's adult FCR uses regional survey data (preferred approach) to provide a more reasonable regional level of protection as compared to EPA guidance and meets the protection level goals for the general public and most populations of fish consumers. Based on the best available survey data, 30 g/d is representative of upper percentile freshwater, caught fish consumption and appropriate for use on a statewide default basis. However, MPCA and EPA have also long recognized that there are individuals and subsistence fishing populations, such as Native American Tribes, that eat more fish for personal and cultural reasons. Therefore, both of

Minnesota's water quality rules have language that allows for the use of higher fish consumption rates when available and reliable data are provided to the Commissioner for developing site-specific modified WQSs (7050.0222, Subps. 2a, 3a, 4a and 8 and 7052.0270). Application of alternate site-specific FCRs could reflect regional fishing practices or subpopulations depending on the available data.

In addition, tribes can be authorized by EPA to develop their own WQSs and two Minnesota Chippewa Tribes in the Lake Superior Basin, the Fond du Lac Band and the Grand Portage Band, have their own WQSs using higher fish consumption rates of 60 g/d and 142.5 g/d, respectively[34]. MPCA continues to review this important topic with EPA and interested parties through the public outreach conducted for WQS Triennial Standards Review and rulemaking.

The revised methods more specifically address the durations of exposure relevant to noncancer health effects and cancer risk. Within this improved consideration of the duration of exposure to a toxicant necessary to elicit a toxic effect or increase cancer risk, the chronic duration, defined as greater than 8 years for noncancer evaluations and 70 years for linear cancer evaluations (54 years being considered as adult exposure with ADAF application), the adult FCR will be applied as the representative chronic FCR. However, if the basis for the chronic adverse effect is specific to developing life stages, whether *in utero* or birth to adulthood, alternate FCRs may be developed in consultation with EPA and MDH.

To match improvements in the way MDH now applies drinking water intakes on a per body weight basis, the FCR will be presented on a per body weight basis in the proposed algorithms. For use in the noncancer and nonlinear cancer chronic and cancer algorithms, the FCR will be expressed as the 30 grams of fish consumed per day divided by the 70 kg adult body weight (still recommended average body weight for males and females, EPA 2000b) as 0.43 grams of fish consumed per kilogram of body weight per day (g/kg-d) with the acronym, FCR_{adult}.

Noncancer toxicants—developmental endpoints and less-than-chronic RfD considerations: Freshwater fish is considered as a potential route of exposure to pollutants in surface water, but the differences in the exposure profile for fish consumption for developmental life stages is quite different from that of drinking water. Two key differences provide the foundation for determining how to apply the children's FCR. First, drinking water is essential. While fish also can provide essential nutrients, they are not a sole source of protein. Second, the age at which most children begin to eat fish is later than the neonatal window included in *Developmental Health Endpoints* and at life stages that generally are shown to be less sensitive. Therefore, the application of the children's FCR in less-than-chronic HH-WQS algorithms or the *Supplemental Algorithm for Developmental Susceptibility* for drinking water (Section V.B.c.) is not supported as a default approach, but would be considered on a pollutant-specific basis.

MPCA considered these factors in evaluating the significance of fish in the diets of infants. Protein and essential amino acids are important components in human diets; however, neonates typically obtain protein from breast milk and formula and generally don't start to eat solid foods, particularly meat and fish, until 6 months to a year of age (USEPA 2008). Even then,

[34] (Links to Tribal Water Quality Standards at http://www.epa.gov/waterscience/standards/wqslibrary/tribes.html#r5)

fish from all sources is only a small percentage of total food intake in the general population (Table 14-9 in USEPA 2008), but can comprise a higher proportion in sport-fishing or Native American households (Tables 10-12 to 10-26 in USEPA 2008). There are many foods that provide essential proteins, yet fish is recognized in the medical community and in many segments of the general population as one of the best, because many species of fish and shellfish are low in fat and are one of the few primary sources of beneficial omega-3 fatty acids that are critical in neurological system development and cardiovascular health. Even if fish is chosen as part of a person's diet, marine, store-bought fish consumption rates are generally much higher than local freshwater sport-caught fish consumption rates (USEPA 2002a, USEPA 2011, Tran et al. 2013, USEPA 2013b).

The application of the children's FCR for ages one through five (to age six)–the ages of higher intake of fish on a per body weight basis as suggested by survey data–has relevance for understanding potential exposure differences over a lifetime; however, given that age window and the profile of defined developmental toxicity (prenatal to neonatal) do not overlap, the children's FCR is not relevant or necessary to apply for all pollutants, but could be used on a pollutant-specific basis when data suggests higher susceptibility during this later life stage. More importantly, when considering the age groups recommended by EPA for risk assessment (USEPA 2005c), and considering developmental protection from pollutants in fish, the subpopulation of women who may become or are pregnant, who are typically defined in the age range of 15 through 44 (USEPA 2000b), is the most relevant.

For this subpopulation of women who may become or are pregnant, MPCA is not developing an alternate fish consumption rate at this time. As part of assessing developmental toxicity, a rate specific to women age 15 to 44 would be reviewed with EPA and MDH when warranted for use on a pollutant-specific basis. In addition, in most cases, for pollutants present in fish that affect the prenatal window, these developmental effects are used as the basis for (e.g., methyl mercury) or are less stringent than their chronic RfDs; therefore, HH-WQSs based on chronic durations are expected to be protective of prenatal effects and other shorter duration health effects characterized for less-than-chronic duration RfDs (see Section V.B.c. regarding drinking water pathway).

As previously discussed, a more meaningful option for including the fish pathway in the context of early life stage protection is to use biomonitoring, coupled with survey data, to better estimate fetal and neonatal exposure to pollutants that accumulate in fish and determine appropriate parameters for accounting for this exposure, which can go beyond just fish intake rates. As was EPA's approach to developing water quality criteria for mercury (USEPA 2001), where fetal and maternal biomonitoring data were used in the development of the RfD; MPCA adopted this criterion as a new fish tissue HH-WQSs in 2008. For other pollutants that bioaccumulate in fish, MPCA will examine use of biomonitoring and survey data reviewed by EPA, MDH, and other credible sources that may assist to better refine exposure parameters for HH-WQSs.

More broadly, application of acute to subchronic RfDs based on any *Health Endpoint* will be examined on a pollutant-specific basis for fish consumption to ensure that the final HH-WQS is protective of effects that can occur at less-than-chronic durations. In addition, if chronic

noncancer effects (greater than 8 years' duration) specifically affect developmental life stages, application of the children's FCR or other alternate rate would be developed with sufficient available and reliable data. These proposed approaches are described again and fully illustrated in Sections V.B. *Noncancer and Nonlinear Carcinogen Algorithms* and *Supplemental Algorithms for Developmental Susceptibility*.

Nonlinear carcinogens:

As described in the section on *Cancer Evaluations,* for carcinogens with nonlinear toxicological profiles, the noncancer algorithms are used; this means use of a RfD for protection from precursor events that can lead to cancer. The studies used for developing the RfD are generally chronic and would mean that the appropriately matched exposure duration will also be "chronic". For this reason, unless pollutant specific data suggest an alternate approach, the appropriate FCR to use will be the chronic FCR_{adult}.

Linear carcinogens:

MPCA is proposing to apply the children's FCR as part of the algorithm for linear carcinogens. The children's FCR is recommended for accounting for exposure differences in children ages one to five. The age ranges that apply for the new cancer potency factor ADAFs for protection of early life stage susceptibility to excess cancer risk applies and is highest for neonates at birth up to 2 years of age ($ADAF_{<2}$ = 10). To ensure the target excess cancer risk does not exceed one in 100,000 for that age range, MPCA will apply the FCR of 0.86 g/kg-d ($FCR_{0 \text{ to } <2}$) without making a time-weighted age-adjustment in order to retain that rate for ages children may start to eat fish (1 to <2 years). For the ADAF age range of 2 to less than 16 years of age ($ADAF_{2 \text{ to } < 16}$ = 3), MPCA will use a time-weighted age-adjusted rate based on applying 0.86 g/kg-d for four years and 0.43 g/kg-d for 10 years for a $FCR_{2 \text{ to } < 16}$ of 0.55 g/kg-d. The final ADAF group is ages 16 to 70 would apply the FCR_{adult}.

When the linear cancer algorithm is based on an $AF_{lifetime}$ with duration of 70 years, adjusting for the higher children's FCR for ages one to five for five years wouldn't significantly increase the existing FCR_{adult} of 0.43 g/kg-d[35]. Therefore, FCR_{adult} will be used when there is an $AF_{lifetime}$ or no cancer potency adjustment.

Alternate Fish Consumption Rates:

As previously described, there are many scenarios where the adult and children's FCRs can be improved upon on a site- or pollutant-specific basis, which were beyond the scope of these revisions. The adult FCR and new children FCRs rates are designed to be applied on a default statewide basis; however, sufficient, available, and reliable data may be available to refine these rates. There may be information available on a pollutant-specific basis, possibly through the use of biomonitoring data, to better refine the appropriate exposure windows relevant to the toxicological profile. There also may be data submitted to develop site- or region-specific FCRs. Any alternate FCR used to develop site-specific criteria or modified standards or future HH-WQS adopted into rule has opportunities for public review and comment as previously described.

[35] Chronic Age-Adjusted FCR using children's FCR for 5 years and adult FCR for 65 years ={[(0.86 g/kg-d x 5) + (0.43 g/kg-d x 65)] / 70} = 0.46 g/kg-d

f. Trophic level distinctions for bioaccumulation factor development

Part of the improvements for human health methods published by EPA in 2000 included new methods, modeled on those developed for the Great Lakes Initiative (GLI), to develop bioaccumulation factors (BAFs) or bioconcentration factors (BCFs) for use in accurately accounting for a pollutant's accumulation in fish tissue (see Section IV.C.g.). Besides using the national CSFII survey data to propose new default fish consumption rates for the general population, the survey data provided information to categorize fish species consumed. EPA used this categorization to propose more accurate estimation of BAFs based on the aquatic food chain trophic levels that the fish and shellfish consumed fit into. From Minn. R. ch. 7052, *Trophic Level* (TL) means:

> the food web level in an ecosystem that is occupied by an organism or group of organisms because of what they eat and how they are related to the rest of the food web. For example, trophic level 3 in an aquatic ecosystem consists of small fish such as bluegills, crappies, and smelt and trophic level 4 consists of larger carnivorous fish such as walleye, salmon, and northern pike.

For nonionic, hydrophobic pollutants and methyl mercury bioaccumulation increases as it moves up trophic levels of the aquatic food chain. TL descriptions used by EPA in the new BAF methods are described in the following table (from USEPA 2003b):

Table 4: Food web structure for national BAF methodology from USEPA 2003b, Table 4-5

Species	Trophic Level	Lipid Content	Weight	Diet
Phytoplankton	1	0.5%		
Zooplankton (mysids [*Mysis relicta]*)	2	5.0%	100 mg	
Benthic Invertebrates (*Diporeia*)	2	3.0%	12 mg	
Sculpin (*Cottus cognatus*)	3	8.0%	5.4 g	18% zooplankton, 82% *Diporeia*
Alewife (*Alosa pseudoharengus*)	3	7.0%	32 g	60% zooplankton, 40% *Diporeia*
Smelt (*Osmerus mordax*)	3-4	4.0%	16 g	54% zooplankton, 21% *Diporeia*, 25% sculpin
Salmonids (*Salvelinus namaycush, Oncorhynchus mykiss, Oncorhynchus velinus namaycush*)	4	11%	2,410 g	10% sculpin, 50% alewife, 40% smelt

As described further in the *Bioaccumulation Factor* section of this document, EPA distinguished general population fish and shellfish consumption as fitting into TLs 2 to 4 and based on the national default recommended fish consumption rate of 17.5 g/d, fell into percentages of

approximately 22% for TL2, 46% for TL3, and 33% for TL4. MPCA is not proposing to use the national default percentages, but will instead apply the TL distinctions recommended in the GLI for Class 2B surface waters for a few key reasons:

1) MPCA's Lake Superior (GLI) and statewide adult fish consumption rate is based on preferred regional data for an important subpopulation of sport-fish consumers;
2) Fish species consumed from Minnesota's lakes and rivers are better defined from the regional surveys, not the national CSFII survey.
3) MPCA has already implemented trophic level distinctions for BAF development in Minn. R. ch. 7052 based on GLI, which used Great Lakes fish consumption data (USEPA 1995a, USEPA 1995d).
4) Use of the GLI trophic level percentages (76% for TL4 and 24% for TL3) better reflects local high consumption of predator (TL4) fish species on inland lakes and rivers, such as walleye, bass, and northern pike (see references in Appendix B1).

The use of the same trophic level percentages for developing statewide HH-WQS as is done for the Lake Superior HH-WQSs also provides consistency in the state and a higher level of protection when calculating *State*-BAFs than using lower trophic level assumptions. In the proposed algorithms (see Sections V.A. and B.), each trophic level BAF will be multiplied by their respective percentages: BAF_{TL3} by 24% and BAF_{TL4} by 76% for Class 2B waters. For Class 2A surface waters protected for cold-water communities and in particular trout, there is no distinction made in TL, because BAF development is based almost exclusively on trout (use of 6% lipid and TL4 bioaccumulation data).

g. Bioaccumulation methods for estimating pollutant concentrations in fish tissue

Introduction

As previously introduced, EPA has published, *Methodology for Deriving Ambient Water Quality Criteria* (AWQC) *for the Protection of Human Health* (2000b). An important factor in the derivation of AWQC is the calculation of BAFs. EPA's 2000 methods made significant improvements to the previous 1980 national BAF methods and MPCA's own methods adopted in 1990. MPCA plans to adopt the 2000 methods with the exception of a few differences where Minnesota or Regional data are used in place of national defaults as described fully in this section. In addition, to remove some of the uncertainty with BAF development, new algorithms are also being proposed for developing HH-WQSs applicable in fish tissue for highly bioaccumulative pollutants. A brief summary of these methods follows.

The goal of EPA 2000 methods for calculating a national BAF (hereafter referred to as *State*-BAF, because the applicable "site" is Minnesota as compared to the nation as referred to in EPA's methods[36]) is to protect humans from long-term exposure to waterborne chemicals by considering long-term, average bioaccumulation potential of a chemical in edible tissues of aquatic organisms, primarily fish. *Bioaccumulation* refers to the uptake and retention of a chemical by an aquatic organism from all surrounding media (e.g., water, food, and sediment).

[36] *State*-BAFs will be furthered refined based on the applicable Class 2 subclass, which distinguishes different fish communities and lipid values (Lake Superior 2A, 2A, and 2B).

Bioconcentration refers to uptake and retention of a chemical by aquatic organisms exposed from water only. Both bioaccumulation and bioconcentration can be viewed as the result of competing rates of chemical uptake and depuration (chemical loss) by an aquatic organism.

BAFs are an important component for accounting for exposure to pollutants through fish consumption when developing protective standards applied as water concentrations (versus fish tissue concentrations that do not require a BAF). As described in detail later, the properties of a pollutant affect whether it is found in an aquatic organism and to what degree. Of greatest concern for human exposure are the pollutants that biomagnify or increase in concentration at each successive level of the aquatic food chain in fish tissue or muscle (the part of the fish most often consumed). When this occurs, HH-WQSs need to be adequately lowered to ensure water concentrations do not reach concentrations over an extended duration (MPCA uses 30 days as the duration for HH-WQSs) that would result in the pollutant reaching levels in fish tissue that could exceed health protective concentrations.

Many of the priority pollutants identified by EPA under the Clean Water Act (CWA) and the main reason behind the Great Lakes Initiative (GLI), basis for Minn. R. ch. 7052, was concern over many *Bioaccumulative Chemicals of Concern* (BCC) being found in fish tissues (e.g., mercury, polychlorinated biphenyls or PCBs and dichlorodiphenyltrichloroethane or DDT) with known impacts to human health and fish-eating birds and wildlife.

> BCCs "means any chemical that has the potential to cause adverse effects which, upon entering the surface waters of the state, by itself or as its toxic transformation product, accumulates in aquatic organisms by a human health bioaccumulation factor (BAF) greater than 1,000, after considering metabolism and other physiochemical properties that might enhance or inhibit bioaccumulation, in accordance with the methodology in part 7052.0110, subpart 3".

As discussed fully in *Strengths and Limitations of BAF Methods*, EPA GLI and 2000 methods are based on best available data, primarily for nonionic organic pollutants that are highly bioaccumulative as described above, and aim to balance the objective of developing average, representative BAFs with the understanding that BAFs differ based on many factors and other classes of chemicals lack detailed data on important BAF processes. MPCA builds on EPA methods and includes Minnesota-specific data to refine the approach for HH-WQSs.

Defining the appropriate scale or "site" for BAFs used in Minnesota's HH-WQSs

The foundation for EPA's national BAF methods comes from the methods developed and adopted by Minnesota as part of the GLI in Minn. R. ch. 7052 for the Lake Superior basin (USEPA 1995a, USEPA 1995d). The GLI BAF methods inherently defined the applicable waters (referred to as "site" in (USEPA 2003b, USEPA 2009) as the "Great Lakes" and as such set default parameters (dissolved and particulate organic carbon or DOC and POC) and trophic level (TL) fish lipid content based on those waterbody characteristics. The CWA and EPA guidance have always recognized that water quality standards are more meaningful and scientifically defensible when based on regional to local data. As described in Section IV.D., MPCA replaced national defaults with regional data for Lake Superior when setting the fish consumption rate and fish lipid value.

In proposing to replace 1990 BAF methods in the statewide methods for HH-WQSs in Minn. R. ch. 7050, MPCA is defining the applicable "site" as Minnesota's or *State* surface waters in place of EPA's 2000 "national" scale. This means the BAF methods will be based on *State* average waterbody characteristics (statewide adult FCR, TL distinction, and organic carbon inputs-POC and DOC) with different fish lipid percentages refining the *State*-BAFs by aquatic life Use Classification: Lake Superior Class 2A (8.5%), Statewide Class 2A-cold water communities and trout fisheries (6%), and Class 2B, cool to warm water communities (TL3 2% and TL4 1.5%).

When developing site-specific Chronic Criteria (CC), the term *State* is replaced with the term *Site*. In this case, the considerations again being the specific Class 2 subclass for fish lipid values and use of the statewide DOC and POC values as default unless site-specific data are available as approved by the commissioner.

Full application of these changes is found in the *Revised Parameters* and *Algorithms* sections.

Strengths and limitations to BAF methods

EPA's BAF methods first address differences in a pollutant's physicochemical characteristics that affect BAF processes. Four Baseline-BAF methods are defined and prioritized based on five main characteristics: organic or inorganic compound, ionization in surface waters, octanol-water partition coefficients (K_{ow}), metabolism in aquatic organisms, and known biomagnification potential. These methods build on the available data and develop appropriate hierarchies of approaches, defined by six Procedures, based on relevant and preferred data for all the chemical classes.

BAF data are most robust for pollutants defined as nonionic organics and have Log K_{ow}s greater than or equal to 4. The strengths of the methods for these pollutants complements the fact that many pollutants of concern in fish tissue fall into this category, except mercury; so for pollutants such as PCBs, dioxins, DDT, and other legacy organochlorine pesticides (dieldrin, chlordane, toxaphene, etc.) there have been substantial field and laboratory datasets and models developed to more accurately define BAFs (EPA 2000b, 2003b and 2009). These richer datasets are reflected in EPA's 2000 BAF methods and subsequent guidance for developing site-specific BAFs (EPA 2009). For these pollutants, the BAF methods can provide very credible average BAFs as explained fully in the following section for application in developing HH-WQS as water concentrations; however, even these pollutants have BAFs that can be variable in a waterbody based on site characteristics, particularly organic carbon parameters, food chains, and resident aquatic species. HH-WQSs are proposed that address these key characteristics by the three Use Classifications for aquatic life communities (Lake Superior-2A, 2A, and 2B); use of other data for smaller, scale definitions of a *site* could be used with EPA approval through the process of "site-specific modification" of a listed standard (Minn. R. 7050.0220, subp. 7 and 7052.0270).

EPA's 2001 AWQC for mercury recognized that BAFs can vary significantly depending on the surface water characteristics. Therefore, to overcome this issue, that mercury criterion was developed and expressed as a fish tissue concentration and not a water concentration. The application of a fish tissue concentration ensures more consistent and meaningful protection for fish consumers. Because fish tissue-based HH-WQSs are more of a direct measure for assessing bioaccumulative pollutants in surface waters and are applied as a single

concentration not to be exceeded in any fish consumed, MPCA plans to supplement the current water concentration CS (or CC) algorithms with algorithms for fish-tissue based standards (CS_{ft}) or site-specific criterion (CC_{ft}) for BCCs as another means to addressing BAF limitations (Section IV.E.).

Conversely, MPCA is refining BAF methods to better address the fact that some classes of pollutants, such as:

- Polycyclic Aromatic Hydrocarbons or PAHs (nonionic organics with Log K_{ow}s > 4 and high metabolism in fish),
- VOCs (e.g., benzene and vinyl chloride) (nonionic organics with Log K_{ow}s < 4),
- Ionic organics, and
- Many metals and metalloids, other than mercury (e.g., inorganic arsenic and cadmium),

often have limited field data and laboratory tissue distribution studies to precisely determine muscle tissue BAFs and chemical forms in that tissue. Many historical BAFs were developed based on K_{ow} or whole fish BCF studies that for many VOCs and PAHs can overestimate exposure from the fish pathway (USEPA 1995a, USEPA 2000b). VOC distribution studies have shown that a chemical such as benzene partitions to lipid-rich organs, such as the gall bladder and liver, but is very quickly eliminated from muscle tissue after exposure ends (Korn et al. 1976). PAHs can accumulate in sediment-dwelling invertebrates, but are metabolized and excreted in fish (USEPA 2000b).

Limited field or toxicokinetic laboratory studies can not only limit precise estimates of bioaccumulation, but also full characterization of the predominate form of the pollutant in tissue. Recent reviews of metal bioaccumulation have found that their partitioning and accumulation in aquatic organisms is very specific to the metal or metalloid and water concentrations, where BAFs were higher when water concentrations were lower (USEPA 2007). Studies on PAHs identify low levels of metabolites, not parent PAHs, in fish tissue, which would be the more appropriate form to use for assessing toxicological profiles (Valdez Domingos et al. 2011).

Overall the best approach for developing BAFs for these types of pollutants–use of field data–is the key data limitation; this also means that actual estimates or measurements of exposure from eating fish when a pollutant of this type is present in a waterbody is lacking to definitively develop a default approach for addressing bioaccumulation and relative significance of the fish pathway in HH-WQSs for these pollutant classes. EPA has approached these data limitations through application of the best available science in their Baseline-BAF Procedures; MPCA plans to adopt EPA's methods, but supplement the BAF review as described below.

Part of addressing data limitations is ensuring the final HH-WQSs are protective and meaningful for addressing surface water pollutants. As described in Section V.I. BAF uncertainty when placed in the context of surface water sources of exposure and use of protective parameters such as RSCs of 0.2 or 0.5 and relatively stringent excess CR levels, would lead to enough conservatism in final values to be protective of human health. Conversely, information available on HH-WQSs should reflect the actual risk posed by the fish pathway in comparison to other

sources of exposure to these pollutants. Therefore, there is a need for specifically providing clearer information that recognizes fish consumption as an insignificant route of actual exposure when warranted or more accurately describing this pathway's relative contribution to a population's total exposure to the pollutant. While there is not enough field data to definitively consider the fish pathway negligible or insignificant on a default basis for some pollutant categories, the details on this route of exposure will be part of each pollutant-specific summary sheets and tables presented when proposing new or revised HH-WQSs into rule. The listing of each pollutant's CS and narrative standards are also being enhanced to reflect these considerations.

MPCA plans to address BAF data limitations and relevance of the fish pathway in three ways:

1) Follow EPA's hierarchy for these pollutant classes, particularly as it relates to a focus on field data and use of supplemental toxicokinetic studies;
2) When sufficient and reliable data are available on a pollutant-specific basis to determine the form and concentration of exposure from the fish pathway, MPCA will use this data to more accurately determine the relevant toxicological data (parent or metabolite in consultation with MDH); and
3) If presence in tissue is insignificant as a route of exposure when considering total environmental exposure to the pollutant, apply the minimal value of "1" as the BAF and note this in the CS listings in Minn. R. chs. 7050 and 7052.

Summary of methods

Separate procedures for deriving *State*-BAFs were developed based on a framework of defining and categorizing general chemical properties: nonionic organic, ionic organic, inorganic, and organometallic (defined in USEPA 2000b, Section 5.3.5). Also, *State*-BAFs are derived separately for two trophic levels (TL): 3 and 4 and using different fish lipid values by Use Classification, to account for differences in fish characteristics and aquatic food webs that affect bioaccumulation.

Please note: The following is an abbreviated summary of the full EPA methods[37]; further explanation and justification of methods can be found in EPA methodology and technical support documents (USEPA 2000b, USEPA 2003b, USEPA 2009).

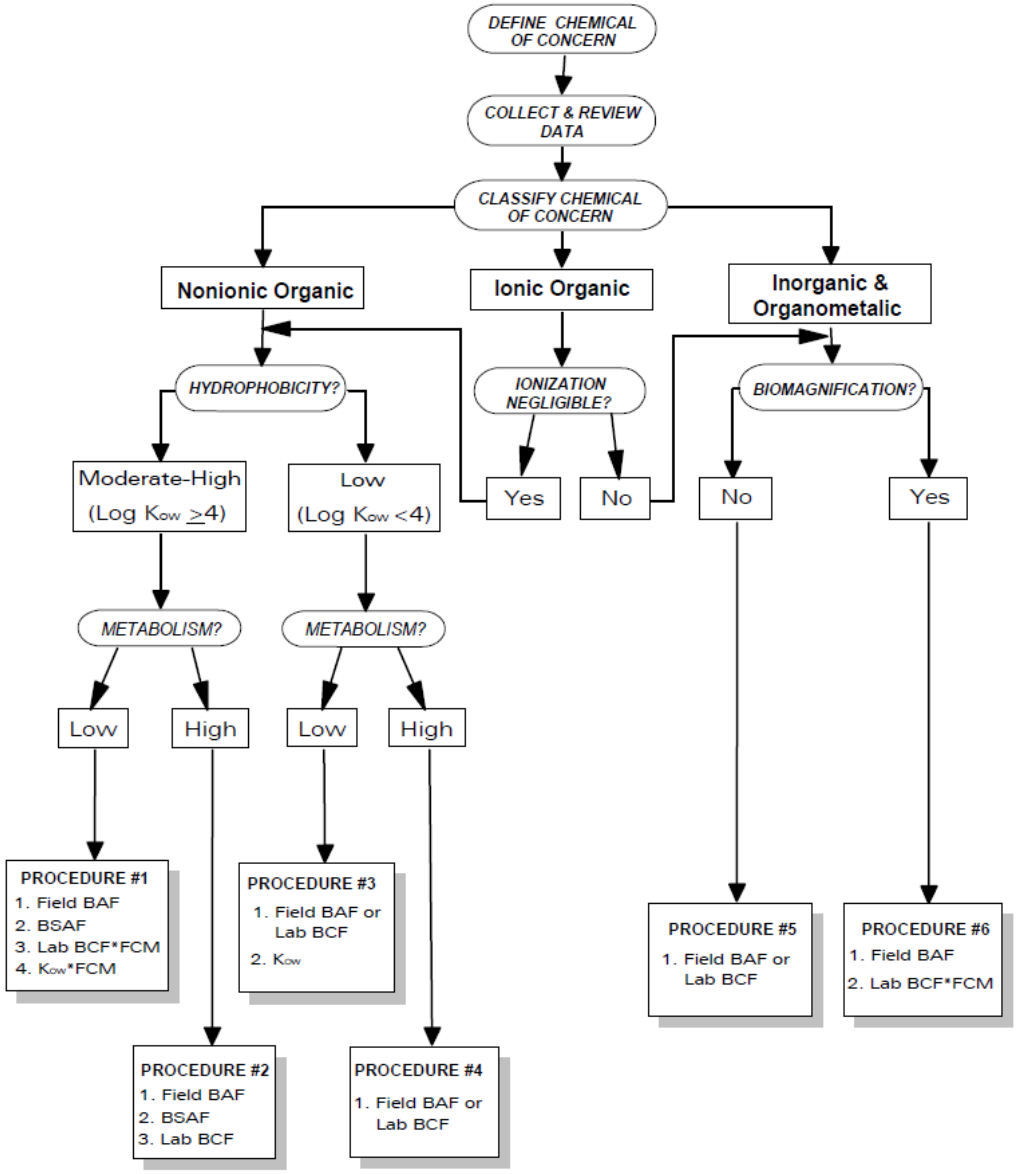
State-BAFs can be measured or predicted using some or all of the following four methods:

1) Measured BAF obtained from a field study;
2) Predicted BAF from field-measured Biota-Sediment Accumulation Factor (BSAF);
3) Predicted BAF from a laboratory-measured Bioconcentration Factor (BCF), with or without adjustment by a food-chain multiplier; and
4) Predicted BAF from a chemical's octanol-water partition coefficient (K_{ow}).

The appropriate method selected for use in developing a Baseline BAF is based on chemical type and physical/chemical (physicochemical) properties according to the framework in Figure 3 (USEPA Figure 5-1) with additional limitations described for in the six available Procedures.

[37] To keep the discussion concise and accurate, some sections were excerpted directly from EPA's 2000 *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health.*

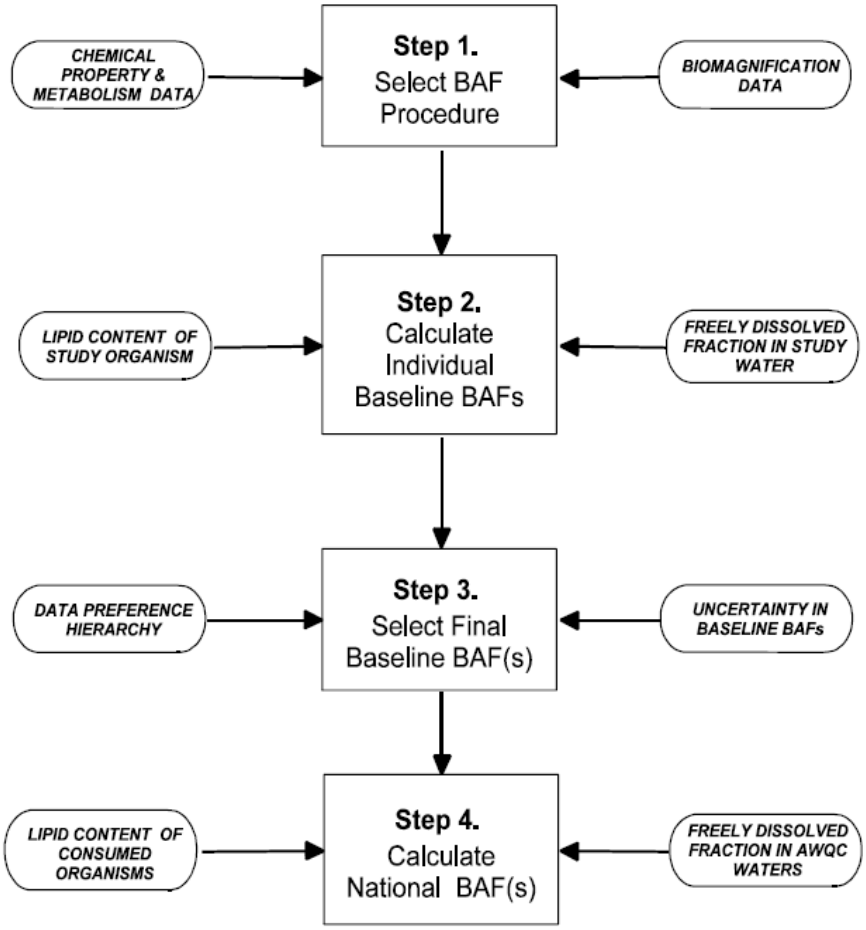
Figure 3: Figure for deriving a national (*State*) BAF from USEPA 2000b, Figure 5-1



For all chemicals there are four general steps for deriving *State*-BAFs as shown in Figure 4 (USEPA):

1) Select the Baseline BAF derivation procedure from the six available
2) Calculate individual Baseline BAFs
3) Select the final Baseline BAFs by trophic level
4) Calculate the *State* BAFs from final Baseline BAFs

Figure 4: BAF derivation for nonionic organic chemicals from USEPA 2000, Figure 5-2



The BAF steps are used to review all available data on a pollutant's bioaccumulative properties (individual Baseline BAFs) to arrive at the most reliable values by TL (final Baseline BAFs) for use with State-specific surface water Use Classifications and fishery data (*State*-BAFs). Once pollutant-specific *State*-BAFs by TL (3 and 4) and Class 2 subclass (input of fish lipid values) are calculated they are used in the derivation of HH-WQSs or site-specific criteria applied in water.

Step 1 of the methodology described above determines which of the BAF procedures described in Figure 3 (USEPA Figure 5-1) will be appropriate for the *State* BAF. Step 2 involves calculating individual, species-specific BAFs using all of the methods available within a selected procedure. Step 3 involves selecting the final Baseline BAF from the individual BAFs by taking into account uncertainty in the individual BAFs and the data preference hierarchy selected in Step 1. Step 4 is the calculation of a BAF that will be used in the derivation of HH-WQSs.

State-BAFs for nonionic organic chemicals

Nonionic organic chemicals, for the purpose of these BAF methods, are organic compounds that do not substantially ionize in ambient waterbodies (USEPA 2000b). These compounds are also referred to as having neutral or nonpolar characteristics, leading to moderate to high lipid or hydrophobicity partitioning in aquatic organisms.

Selecting a BAF derivation procedure for nonionic organic chemicals

As shown in Figure 3 (USEPA Figure 5-1), two decision points exist in selecting a BAF derivation procedure. The first decision point requires knowledge of a chemical's hydrophobicity (K_{ow}). Chemicals with a log $K_{ow} \geq 4$ are considered moderately to highly hydrophobic and may bioaccumulate in aquatic food webs. The second decision point is based on the rate of metabolism for the chemical in the target organism.

Procedure #1 should be used to derive *State*-BAFs for moderately to highly hydrophobic nonionic organic chemicals in cases where:

a) the rate of chemical metabolism by the target aquatic organism is expected to be sufficiently low such that biomagnification is of concern, or
b) the rate of chemical metabolism by target aquatic organism is not sufficiently known.

Procedure #1 accounts for non-aqueous exposure and the potential for biomagnification in aquatic food webs through the use of field-measured values for bioaccumulation (i.e., field-measured BAF or BSAF) and food chain multipliers (FCMs) when appropriate field data are unavailable.

Examples of nonionic organic chemicals for which Procedure #1 is appropriate:

Chlorobenzenes, PCBs, Dieldrin, DDT, Heptachlor, and Chlordane

Procedure #2 should be used to derive the *State*-BAFs for moderately to highly hydrophobic nonionic organic chemicals in cases where:

a) the rate of chemical metabolism by target aquatic organisms is expected to be sufficiently high such that biomagnification is not of concern.

   Procedure #2 relaxes the requirement of using FCMs and eliminates the use of K_{ow}-based estimates of the BAF, two procedures that are most appropriate for poorly metabolized nonionic organic chemicals.

   Nonionic organic chemicals with log K_{ow} values < 4 should be classified as exhibiting low hydrophobicity. For these chemicals, non-aqueous exposure is not likely to be important in determining chemical residues in aquatic organisms.

Procedure #3 should be used to derive *State*-BAFs for nonionic organic chemicals of low hydrophobicity in cases where:

a) the rate of chemical metabolism by target aquatic organisms is expected to be negligible, such that tissue residues of the chemical of concern are not substantially reduced compared to an assumption of no metabolism, or
b) the rate of chemical metabolism by target aquatic organisms is not sufficiently known.

Procedure #3 includes the use of K_{ow}-based estimates of the BCF to be used when lab or field data are absent.

Procedure #4 should be used to derive *State*-BAFs for nonionic organic chemicals of low hydrophobicity in cases where:

a) the rate of chemical metabolism by target aquatic organisms is expected to be sufficiently high, such that tissue residues of the chemical of concern are substantially reduced compared with an assumption of no metabolism.

Procedure #4 eliminates the option of using K_{ow}-based estimates of the BAF because the K_{ow} may over-predict accumulation when a chemical is metabolized substantially by an aquatic organism.

Calculating individual Baseline BAFs

Calculating an individual Baseline BAF involves normalizing the field-measured BAF (or lab-measured BCF) which are based on total concentration in tissue and water, by the lipid content of the study organism and the freely dissolved concentration in the study water.

1. For each species for which acceptable data are available[38], calculate all possible Baseline BAFs using each of the following four Methods: (1) Measured BAF obtained from a field study, (2) BAF predicted from field-measured BSAF, (3) BAF predicted from a laboratory-measured BCF (with or without adjustment by a FCM), (4) BAF predicted from a chemical's octanol-water partition coefficient (K_{ow}).
2. Individual baseline BAFs should be calculated from field-measured BAFs, field-measured BSAFs, lab BCFs, and the Kow according to the following procedures.

Method 1: *Baseline BAF_{fd}s from field-measured BAFs*

$$Baseline\ BAF_l^{fd} = \left[\frac{Measured\ BAF_T^t}{f_{fd}}\right]\left[\frac{1}{f_l}\right] \qquad \text{(EPA Equation 5-10)}$$

Where:

Baseline BAF_l^{fd} = BAF expressed on a freely dissolved and lipid-normalized basis (L/kg)

Measured BAF_T^t = BAF based on total concentration in tissue and water (L/kg)

f_l = Fraction of the tissue that is lipid

f_{fd} = Fraction of the total chemical that is freely dissolved in ambient water

Determining the measured BAF_T^t

[38] BCF and BAF data are most relevant from TL3 and 4 fish species; however, data available from lower TL fish species and other aquatic organisms will be reviewed for qualitative evaluation to improve the BAF evaluation and application of TL 3 and 4 *state*-BAFs.

The field-measured BAF should be calculated based on total concentration of the chemical in the appropriate tissue of the aquatic organism and the total concentration of the chemical in the ambient water at the site of the sampling. The equation to derive a measured BAF_T^t is:

$$Measured\ BAF_T^t = \frac{C_t}{C_w} \qquad \text{(EPA Equation 5-11)}$$

Where:

C_t = Total concentration of the chemical in the specified wet tissue (µg/kg)

C_w = Total concentration of chemical in water (µg/L)

Determining the fraction freely dissolved (f_{fd})

The fraction of the nonionic organic chemical that is freely dissolved in the study water is required for calculating a baseline BAF_{fd} from a field-measured BAF_T^t. The freely dissolved fraction is the portion of the nonionic organic chemical that is not bound to particulate organic carbon (POC) or dissolved organic carbon (DOC).

$$f_{fd} = \frac{1}{[1 + (POC \times K_{OW}) + (DOC \times 0.08 \times K_{OW})]} \qquad \text{(EPA Equation 5-12)}$$

Where:

POC = Concentration of particulate organic carbon (kg/L)

DOC = Concentration of dissolved organic carbon (kg/L)

K_{OW} = Octanol water partition coefficient for the chemical

POC and DOC concentrations should be obtained from the original study from which the field-measured BAF is determined. If POC and DOC concentrations are not reported in the BAF study, reliable estimates of POC and DOC might be obtained from other studies closely related to the site(s) or within the same waterbody. If no site specific data are available, EPA national default DOC and POC values will be used as they are generally representative of average surface water conditions (DOC of 2.9 mg/L; POC of 0.5 mg/L, converted to kg/L by dividing by 1,000,000) (USEPA 2003b).

Method 2: *Baseline BAF_l^{fd} derived from BSAFs*

$$\left(Baseline\ BAF_l^{fd}\right)_i = (BSAF)_i \frac{(D_{i/r})(\tilde{O}\ socw)_r (K_{OW})_i}{(K_{OW})_r} \qquad \text{(EPA Equation 5-14)}$$

Where:

$\left(Baseline\ BAF_l^{fd}\right)_i$ = BAF expressed on a freely dissolved and lipid-normalized basis for chemical of interest "i" (L/Kg)

$\left(BSAF\right)_i$ = Biota-sediment accumulation factor for chemical of interest "i" (kg organic carbon/kg of lipid)

$\left(\tilde{O}\ socw\right)_r$ = Sediment organic carbon to freely dissolved concentration ratio of reference chemical "r" (L/kg sediment organic carbon)

$\left(K_{OW}\right)_i$ = Octanol-water partition coefficient for the chemical of interest "i"

$\left(K_{OW}\right)_r$ = Octanol-water partition coefficient for the reference chemical "r"

$\left(D_{i/r}\right)$ = Ratio between $\prod_{socw}$/ K_{ow} for chemicals "i" and "r" (normally chosen so that $D_{i/r} = 1$)

Determining field-measured BSAFs

$$BSAF = \frac{C_l}{C_{soc}} \qquad \text{(EPA Equation 5-15)}$$

Where:

C_l = lipid-normalized concentration of a chemical in an organism (µg/g lipid)

C_{soc} = organic-carbon normalized concentration of a chemical in surface sediment samples (µg/g sediment organic carbon)

Lipid-normalized concentration

$$C_l = \frac{C_t}{f_l} \qquad \text{(EPA Equation 5-16)}$$

Where:

C_t = concentration of the chemical in the wet tissue (either whole organism or specified tissue) (µg/g)

f_l = fraction lipid content in the tissue

Organic-carbon normalized concentration

$$C_{soc} = \frac{C_s}{f_{oc}} \qquad \text{(EPA Equation 5-17)}$$

Where:

C_s = concentration of chemical in dry sediment (µg/g sediment)

f_{oc} = fraction organic carbon in dry sediment

Sediment-to-water partition coefficient

$$(\tilde{O}_{socw})_r = \frac{(C_{soc})_r}{(C_w^{fd})_r} \qquad \text{(EPA Equation 5-18)}$$

Where:

$(C_{soc})_r$ = concentration of the reference chemical "r" in dry sediment normalized to sediment organic carbon (µg/kg sediment organic carbon)

$(C_w^{fd})_r$ = concentration of the reference chemical "r" freely dissolved in water (µg/L) – freely dissolved fraction calculation uses equations from Method 1 and study-specific or default POC and DOC values.

Selecting reference chemicals

Reference chemicals with $(\prod_{socw})_r / (K_{ow})$ similar to that of the chemical of interest are preferred for this method; complete guidance on this selection is available in EPA 2003.

Method 3*: Baseline* $Baseline\ BAF_l^{fd}$ *from a laboratory-*$Measured\ BCF_T^t$ *and FCM*

The BCF_T^t is used in conjunction with a FCM, because non-aqueous routes of exposure and subsequent biomagnification are of concern for the types of chemicals applicable to Procedure #1.

Baseline BAF_l^{fd} *Equation*

$$Baseline\ BAF_l^{fd} = (FCM)\left[\frac{Measured\ BCF_T^t}{f_{fd}} - 1\left(\frac{1}{f_l}\right)\right] \qquad \text{(EPA Equation 5-19)}$$

Where:

Baseline BAF_l^{fd} = BAF expressed on a freely dissolved and lipid-normalized basis (L/kg)

$Measured\ BCF_T^t$ = BCF based on total concentration in tissue and water (L/kg)

f_l = Fraction of the tissue that is lipid

f_{fd} = Fraction of the total chemical in the test water that is freely dissolved; POC and DOC (or TOC based on data that POC is typically not detected) values measured in the test water are used, unless not available, then the following defaults are used based on typical lab water characteristics: DOC of 2.5 mg/L and POC at 0 mg/L, converted to kg/L by dividing by 1,000,000.

FCM = Food chain multiplier either obtained from Table 5 (USEPA 2000b, Table 5-1 or for a wider range of Log K_{ow} values, USEPA 1995, Table 2 or 40 CFR 132, Appendix B) by linear interpolation for the appropriate trophic level, or from available and reliable field studies.

Determining the measured BCF

$$Measured\ BCF_T^t = \frac{C_t}{C_w} \qquad \text{(EPA Equation 5-20)}$$

Where:

C_t = total concentration of the chemical in the specified wet tissue (µg/kg)

C_w = total concentration of chemical in the laboratory test water (µg/L)

Table 5: Food-chain multipliers for trophic levels 2, 3, and 4 from USEPA 2000b, Table 5-1

Food-Chain Multipliers for Trophic Levels 2, 3 and 4
(Mixed Pelagic and Benthic Food Web Structure and $\bullet_{socw} / K_{OW} = 23$)

Log K_{OW}	Trophic Level 2	Trophic Level 3	Trophic Level 4	Log K_{OW}	Trophic Level 2	Trophic Level 3	Trophic Level 4
4.0	1.00	1.23	1.07	6.6	1.00	12.9	23.8
4.1	1.00	1.29	1.09	6.7	1.00	13.2	24.4
4.2	1.00	1.36	1.13	6.8	1.00	13.3	24.7
4.3	1.00	1.45	1.17	6.9	1.00	13.3	24.7
4.4	1.00	1.56	1.23	7.0	1.00	13.2	24.3
4.5	1.00	1.70	1.32	7.1	1.00	13.1	23.6
4.6	1.00	1.87	1.44	7.2	1.00	12.8	22.5
4.7	1.00	2.08	1.60	7.3	1.00	12.5	21.2
4.8	1.00	2.33	1.82	7.4	1.00	12.0	19.5
4.9	1.00	2.64	2.12	7.5	1.00	11.5	17.6
5.0	1.00	3.00	2.51	7.6	1.00	10.8	15.5
5.1	1.00	3.43	3.02	7.7	1.00	10.1	13.3
5.2	1.00	3.93	3.68	7.8	1.00	9.31	11.2
5.3	1.00	4.50	4.49	7.9	1.00	8.46	9.11
5.4	1.00	5.14	5.48	8.0	1.00	7.60	7.23
5.5	1.00	5.85	6.65	8.1	1.00	6.73	5.58
5.6	1.00	6.60	8.01	8.2	1.00	5.88	4.19
5.7	1.00	7.40	9.54	8.3	1.00	5.07	3.07
5.8	1.00	8.21	11.2	8.4	1.00	4.33	2.20
5.9	1.00	9.01	13.0	8.5	1.00	3.65	1.54
6.0	1.00	9.79	14.9	8.6	1.00	3.05	1.06
6.1	1.00	10.5	16.7	8.7	1.00	2.52	0.721
6.2	1.00	11.2	18.5	8.8	1.00	2.08	0.483
6.3	1.00	11.7	20.1	8.9	1.00	1.70	0.320
6.4	1.00	12.2	21.6	9.0	1.00	1.38	0.210
6.5	1.00	12.6	22.8				

Method 4: *Baseline BAF_l^{fd} from a K_{ow} and FCM*

In this method, K_{ow} is assumed to be equal to the baseline BCF_{fd}. Numerous investigations have demonstrated a linear relationship between log BCF and log Kow for organic chemicals found in fish and other aquatic organisms.

$$Baseline\ BAF_l^{fd} = (FCM) \times (K_{OW}) \qquad \text{(EPA Equation 5-27)}$$

Where:

Baseline BAF_l^{fd} = BAF expressed on a freely dissolved and lipid-normalized basis (L/kg)

FCM = Food chain multiplier either obtained from Table 5 (USEPA 2000b, Table 5-1 or for a wider range of Log K_{ow} values, USEPA 1995, Table 2 or 40 CFR 132, Appendix B) by linear interpolation for the appropriate trophic level, or from available and reliable field studies.

K_{OW} = octanol-water partition coefficient

The BCF-K_{ow} relationship has been developed primarily for nonionic organic chemicals that are not readily metabolized by aquatic organisms and thus is most appropriate for poorly-metabolized nonionic organic chemicals with large log K_{ow}s (i.e., > 6).

Selecting final baseline BAF_l^{fd} *s*

After calculating individual baseline BAF_l^{fd} s using as many of the methods in Procedure #1 as possible, the next step is to determine the final baseline BAF_l^{fd} s for each trophic level from the individual baseline BAF_l^{fd} s. The final baseline BAF_l^{fd} will be used in the last step to determine the *State*-BAF for each trophic level. The final baseline BAF_l^{fd} for each trophic level should be determined from the individual baseline BAF_l^{fd} using the data preference hierarchy in Procedure #1. The following steps and guidelines should be followed for selecting the final baseline BAF_l^{fd} using Procedure #1.

1. **Calculate species-mean baseline BAF_l^{fd} s.** For each BAF method where more than one acceptable baseline BAF_l^{fd} is available for a given species, calculate a species-mean baseline BAF_l^{fd} as the geometric mean of all available individual baseline BAF_l^{fd} s. Highly uncertain BAFs should not be used in calculating a mean. Large differences in individual baseline BAF_l^{fd} s for a given species (e.g., greater than a factor of 10) should be investigated further.
2. **Calculate trophic-level-mean baseline BAF_l^{fd} s.** For each BAF method where more than one acceptable species-mean baseline BAF_l^{fd} is available within a given trophic level, calculate a trophic-level mean baseline BAF_l^{fd} as the geometric mean of acceptable species-mean baseline BAF_l^{fd} s in that trophic level.
3. **Select a final baseline BAF_l^{fd} for each trophic level.** For each trophic level, select the final baseline BAF_l^{fd} using best professional judgment by considering: (1) the data preference hierarchy shown previously, (2) the relative uncertainty in the trophic-level-mean baseline BAF_l^{fd} s derived using different methods, and (3) the weight of evidence among the four methods.

Calculating State-BAFs

The last step in deriving *State*-BAFs for each TL is to convert the final baseline BAF_l^{fd} determined in the previous step to a BAF that reflects conditions of the surface waters by Use Classification to where the HH-WQSs will apply. Converting a final baseline BAF_l^{fd} to *State*-BAFs requires information on: (1) the percent lipid of the aquatic organisms commonly consumed by humans, and (2) the freely dissolved fraction of the chemical of concern that would be expected in the ambient waters of interest. For each TL, a *State*-BAF should be determined from a final baseline BAF_l^{fd} according to the following guidelines.

State-BAF equation. For each TL (3 and 4), calculate a *State*-BAF using the following equation.

$$State\ BAF_{(TL\ n)} = \left[\left(Final\ Baseline\ BAF_l^{fd}\right)_{TL\ n} \times \left(f_l\right)_{TL\ n} + 1\right] \times \left(f_{fd}\right)$$

State BAF Equation 1

Where:

$\left(Final\ Baseline\ BAF_l^{fd}\right)_{TL\ n}$ = Final trophic-level-mean baseline BAF expressed on a freely dissolved and lipid-normalized basis for trophic level "*n*" (L/kg)

$\left(f_l\right)_{TL\ n}$ = Lipid fraction of aquatic species consumed at trophic level "*n*"

f_{fd} = Fraction of the total chemical in water that is freely dissolved

State default lipid values

For existing HH-WQSs parameters, MPCA had previously derived their own fish lipid values for use in Lake Superior, cold water trout fisheries (Class 2A) and cool and warm water fisheries (Class 2B,2C,2D). MPCA adopted an 8.5% lipid value for use in the Lake Superior Basin for Class 2A in Minn. R. ch. 7052 in place of the default GLI values of 1.82% and 3.10% for TLs 3 and 4, respectively (USEPA 1995a). MPCA recognized the popularity of fishing for lake trout and other salmonid species in Lake Superior and their higher lipid content. For that same reason, MPCA developed a trout-based lipid content in 1990 of 6 % for use in Class 2A HH-WQSs. Trout are the keystone species and reason for water being designated Class 2A waters, so the use of a higher lipid value representative of those species of fish is still appropriate and more meaningful than EPA defaults arrived at from national fish surveys (USEPA 2000b).

With the addition of TL distinctions statewide and small dataset used for the Class 2B fish lipid derivation (Appendix B3), MPCA did examine fish lipid content as part of adopting the new BAF methods. Fully described in Appendix A5, more Minnesota-specific data are now available from the interagency Fish Contaminant Monitoring Program (FCMP) database. This fish lipid data is the basis for the new Class 2B TL3 fish lipid value of 2% and continued use of 1.5% for TL4 BAFs. These values meet the goals of developing average BAFs for use in HH-WQSs.

Freely dissolved fraction

Determination of the f_{fd} is based on EPA Equation 5-12 and *State* or GLI values for DOC and POC in ambient surface waters of Lake Superior and statewide inland lakes and streams, respectively. The DOC and POC values used for Lake Superior HH-WQSs are already in Minn. R. ch. 7052, 2 mg/L and 0.04 mg/L, respectively. As part of the BAF methods revision for Minn. R. ch. 7050, MPCA developed representative values to use statewide. As described fully in Appendix A4, the DOC value is 7.5 mg/L is used for DOC and 0.5 mg/L for POC. The DOC is based on the lowest median from representative Minnesota lake and stream datasets; POC value is the national default from EPA (2000b);

Deriving *State*-BAFs using Procedure #1

The types of nonionic organic chemicals for which Procedure #1 is most appropriate are those that are classified as **moderately to highly hydrophobic** and **subject to low (or unknown) rates of metabolism** by aquatic biota. The following four methods can be used in deriving a state BAF:

1) Using a BAF from an acceptable field study (i.e., a field-measured BAF)
2) Predicting a BAF from an acceptable field-measured BSAF
3) Predicting a BAF from an acceptable laboratory-measured BCF and FCM
4) Predicting a BAF from an acceptable K_{ow} and FCM

Because this type of pollutant has biomagnification properties, a FCM is used in relationship to the methods that use BCF or K_{ow} measures, because they do not account for an increase in each step up the food chain or trophic level.

After selecting a derivation procedure, the next steps are: (1) calculating individual baseline BAF_l^{fd} s, (2) selecting the final baseline BAF_l^{fd} s, and (3) calculating the *State*-BAFs. As stated previously, preference in the hierarchy of the available data always points to field data when available.

Deriving *State*-BAFs using Procedure #2

The types of nonionic organic chemicals for which Procedure #2 is most appropriate are those that are classified as **moderately to highly hydrophobic** and **subject to high rates of metabolism** by aquatic biota. The following three methods can be used in deriving a state BAF:

5) Using a BAF from an acceptable field study (i.e., a field-measured BAF)
6) Predicting a BAF from an acceptable BSAF
7) Predicting a BAF from an acceptable BCF

Each of these three methods relies on measured data for assessing bioaccumulation and therefore, includes the effects of chemical metabolism by the study organism in the BAF estimate. Because biomagnification is not an overriding concern for nonionic organic chemicals applicable to procedure #2, FCMs are not used in the derivation of a baseline BAF_l^{fd} from a laboratory-measured BCF_T.

After selecting a derivation procedure, the next steps are: (1) calculating individual baseline BAF_l^{fd} s, (2) selecting the final baseline BAF_l^{fd} s, and (3) calculating the *State*-BAFs.

Deriving *State*-BAFs using Procedure #3

The types of nonionic organic chemicals for which Procedure #3 is most appropriate are those that are classified as **low in hydrophobicity (i.e., log K_{ow} < 4)** and **subject to low (or unknown) rates of metabolism** by aquatic biota. Because biomagnification is not an overriding concern for nonionic organic chemicals applicable to Procedure #3, FCMs are not used in this procedure.

The following three methods can be used in deriving a *State*-BAF:

1) Using a BAF from an acceptable field study (i.e., a field-measured BAF)
2) Predicting a BAF from an acceptable laboratory-measured BCF
3) Predicting a BAF from an acceptable K_{ow}

After selecting a derivation procedure, the next steps are: (1) calculating individual baseline BAF_l^{fd} , (2) selecting the final baseline BAF_l^{fd} , and (3) calculating the *State*-BAFs.

Due to their low hydrophobicity, a freely dissolved fraction of 1.0 should be assumed for calculating *State*-BAFs for nonionic organic chemicals using Procedure #3.

Deriving *State*-BAFs using Procedure #4

The types of nonionic organic chemicals for which Procedure #4 is most appropriate are those that are classified as having **low hydrophobicity** and **subject to high rates of metabolism** by aquatic biota. FCMs are not used in this procedure. Also, K_{ow}-based predictions of bioconcentration are not used in this procedure since the K_{ow}/BCF relationship is primarily based on poorly metabolized chemicals. The following two methods can be used in deriving a *State* BAF:

1) Using a BAF from an acceptable field study (i.e., a field-measured BAF)
2) Predicting a BAF from an acceptable BCF

After selecting a derivation procedure, the next steps are: (1) calculating individual baseline BAF_l^{fd} , (2) selecting the final baseline BAF_l^{fd} , and (3) calculating the *State*-BAFs.

Due to their low hydrophobicity, a freely dissolved fraction of 1.0 should be assumed for calculating *State*-BAFs for nonionic organic chemicals using Procedure #4.

***State*-BAFs for ionic organic chemicals**

Ionic organic chemicals contain functional groups which can either readily donate or exchange protons (e.g., organic acids with hydroxyl, carboxylic, and sulfonic groups) or readily accept protons (e.g., organic bases with amino and aromatic heterocyclic nitrogen groups). Some examples of ionic organic compounds include:

- Perfluorochemicals (e.g., perfluorooctanoic acid or PFOA and perfluorooctane sulfonic acid or PFOS)
- chlorinated phenols (e.g., 2,4,6-trichlorophenol and pentachlorophenol)

- nitrophenols
- aliphatic and aromatic amines (e.g., trimethylamine and aniline)
- linear alkylbenzenesulfonates (LAS) surfactants.

Ionic organic chemicals are considered separately for deriving *State*-BAFs because the anionic or cationic species of these chemicals behave much differently in the aquatic environment compared with their neutral (un-ionized) counterparts. The neutral species of ionic organic chemicals are thought to behave in a similar manner as nonionic organic compounds (e.g., partitioning to lipids and organic carbon as a function of hydrophobicity). However, ionized (anionic, cationic) species exhibit a considerably more complex behavior involving multiple environmental partitioning mechanisms (e.g., ion exchange, electrostatic, and hydrophobic interactions) and a dependency on pH and other factors including ionic strength and compositions. As a consequence, methods to predict the environmental partitioning of organic cations and anions are less developed and validated compared with methods for nonionic organic chemicals.

Procedures for deriving *State*-BAFs for these chemicals differ depending on the extent to which the fraction of the total chemical is likely to be represented by the ionized species in ambient surface waters; this is a function of the ionization properties of the pollutant, measured as pK_a, or the acid-base dissociation constant expressed as the negative $\log_{10}$, and pH of the waterbody. When a significant fraction of the total chemical is expected to be present as the ionized species in water, procedures for deriving the *State*-BAFs rely on empirical methods **(Procedures #5 and #6)**. When an insignificant or minimal fraction of the total chemical is expected to be present as the ionized species (i.e., the chemical exists mostly in neutral form), procedures for deriving the *State*-BAFs will follow those established for nonionic organic chemicals (#1-4). The following guidelines apply for assessing the occurrence of cationic and anionic forms at typical surface water pH ranges for categorization of a chemical as ionic organic.

- MPCA regularly includes pH measurements with surface water monitoring. Average values range from 7.2 to 9 (see summary values by basin at http://www.pca.state.mn.us/wfhye42); in addition, Minnesota's WQSs set minimum and maximum pH values of 6.5 and 8.5, respectively, for most Class 2 surface waters.
- For the ionic organic chemicals, compare pK_a to the range of pH values expected in Minnesota surface waters. At a pH equal to pK_a, 50% of the organic acid or base is expected to be present in the ionized species.
- For organic acids, the chemical will exist almost entirely in its un-ionized form when pH is about 2 or more units below the pK_a. For organic bases, the chemical will exist almost entirely in its un-ionized form when pH is about 2 or more units above the pK_a. In these cases, the chemical would fit the categorization of nonionic organic.
- In ambient waters, most organic acids exist primarily in ionized form and most organic bases in un-ionized forms.
- When pH is greater than pK_a minus 2 for organic acids (or less than pK_a plus 2 for organic bases), the fraction of the total chemical that is expected to exist in its ionized

form can become significant. In these cases, the chemical would be categorized with inorganic and organometallic chemical and follow Procedures #5 or #6 for development of a *State*-BAF.

- Since pH is a controlling factor for dissociation and subsequent partitioning of ionic organic chemicals, consideration should be given to expressing BAFs or BCFs as a function of pH where sufficient data exist to reliably establish such relationships.

State-BAFs for inorganic and organometallic chemicals

The inorganic and organometallic chemicals include inorganic minerals, other inorganic compounds and elements, metals (e.g. copper, cadmium, chromium, zinc), metalloids (selenium, arsenic), and organometallic compounds (e.g. methylmercury, tributyltin, tetraalkyllead) (USEPA 2000b). The derivation of BAFs for inorganic and organometallic chemicals differs in several ways from procedures for nonionic organic chemicals. First, lipid normalization of chemical concentrations in tissues does not generally apply for inorganic and organometallic compounds. Thus, BAFs and BCFs cannot be extrapolated from one tissue to another based on lipid-normalized concentrations of nonionic organic chemicals. Second, the bioavailability of inorganics and organometallics in water tends to be chemical-specific and thus, the techniques for expressing concentrations of nonionic organic chemicals based on freely dissolved form do not apply. Third, at the present time there are no generic bioaccumulation models that can be used to predict BAFs for inorganic and organometallic chemicals as a whole, unlike the existence of K_{ow}-based models for nonionic organic chemicals. While some chemical-specific bioaccumulation models have been developed for inorganic and organometallic chemicals, those models currently tend to require site-specific data for input to the model and are restricted to site-specific applications.

State-BAFs can be derived using two procedures **(#5 and #6)** for inorganic and organometallic chemicals. The choice of BAF derivation procedure depends on whether or not the chemical undergoes substantial biomagnification in aquatic food webs, defined as BAFs or BMFs greater than 1,000.

- For many inorganic and organometallic chemicals, biomagnification does not occur and BCF will be equal to BAF. For these types of chemicals, Procedure #5 should be used to derive the *State*-BAF.
- For some inorganic and organometallic chemicals (e.g., methylmercury), biomagnification does occur and Procedure #6 should be used to derive the *State*-BAF.

The chemical-specific nature of inorganic and organometallic bioavailability is likely due in part to chemical-specific differences in several factors which affect bioavailability and bioaccumulation. These factors include differences in the mechanisms for chemical uptake by aquatic organisms, differences in sorption affinities to biotic and abiotic ligands, and differences in chemical speciation in water. Some inorganic and organometallic chemicals exist in multiple forms and valence states in aquatic ecosystems that can differ in their bioavailability and undergo conversion between forms. The occurrence and bioavailability of different forms of these chemicals should be carefully considered when deriving *State*-BAFs.

- If data indicate that: (1) a particular form (or multiple forms) of the chemical of concern largely govern its bioavailability to target aquatic organisms, and (2) BAFs are more reliable when derived using bioavailable form(s) compared with using other form(s) of the chemical of concern, then BAFs and BCFs should be based on appropriate bioavailable forms.
- Because different forms of many inorganic and organometallic chemicals may interconvert once released to the aquatic environment, regulatory and mass balance considerations typically require an accounting of total concentration in water. In these cases, sufficient data should be available to enable conversion between total concentrations and the other (presumable more bioavailable) forms in water.

Deriving BAFs using procedure #5

In Procedure #5, two methods are available and of equal preference to derive the *State*-BAFs for a given trophic level:

1) Using a BAF from an acceptable field study (i.e., field-measured BAF)
2) Predicting a BAF from an acceptable laboratory-measured BCF

Conversion of field-measured BAFs to baseline BAF_l^{fd} based on lipid-normalized and freely-dissolved concentrations **does not** apply for inorganic and organometallic chemicals. An analogous procedure in concept might be required for converting total BAFs to BAFs based on the most bioavailable form(s) for some inorganic and organometallic chemicals. In addition:

- BAFs should be expressed on a wet-weight basis.
- BAFs should be based on concentrations in the edible tissue(s) of the biota unless it is demonstrated that whole-body BAFs are similar to edible tissue BAFs.
- The concentrations of an inorganic or organometallic chemical in a bioaccumulation study should be greater than normal background levels and greater than levels required for normal nutrition of the test species if the chemical is a micronutrient, but below levels that adversely affect the species.
- Trophic level differences should be evaluated, but may not be relevant for these chemicals and all species data can be combined for a single trophic level BAF.

Deriving BAFs using procedure #6

The types of inorganic an organometallic chemical for which Procedure #6 is appropriate are those that are considered likely to biomagnify in aquatic food webs. Two methods are available to derive the *State*-BAFs:

1) Using a BAF from an acceptable field study (i.e., field-measured BAF)
2) Predicting a BAF from an acceptable laboratory-measured BCF and a FCM

Deriving *State*-BAFs using procedures #5 or #6

The baseline BAF_T^t using total chemical concentrations or bioavailable form(s) directly applied as the *Site*-BAF (fish lipid and freely dissolved fraction adjustments are not needed):

$$Site\ BAF_{(TL\ n)} = Final\ Baseline\ BAF_{(TLn)}$$

Deriving *Site*-BAFs for use in Site-specific Criteria

The methods and procedures for developing a *Site*-BAF are the same as those used for developing *State*-BAFs, except site-specific data can be used in place of state default values if data are determined to meet available data specifications as approved by the commissioner. Application of *Site*-BAFs would be reviewed as described for use of site-specific criteria in Minn. R. ch. 7050 or 7052 for application in the Lake Superior Basin.

h. The Relative Source Contribution factor: accounting for additional exposure

The Relative Source Contribution (RSC) is the fraction of total allowable daily dose of a toxic pollutant that is attributed to drinking water and fish consumption relative to other sources of pollutant exposure to humans, such as air or food (Minn. R. ch. 7050 definition). MDH, in their HRL rule (Minn. R. ch. 4717), defines the RSC as a factor used in drinking water risk assessment to allocate only a portion of the RfD to exposure from ingestion of water, and reserves the remainder of the RfD for other exposures, such as exposures from non-ingestion routes of exposure to water (e.g., inhalation of volatilized chemicals or dermal absorption), as well as, exposures via other contaminated media, such as food, air, and soil (MDH 2008a). The purpose of the RSC is to ensure that the level of the chemical allowed by a HH-WQS, when combined with other identified sources of exposure common to the population of concern, will not result in exposures that exceed the RfD (USEPA 2000b). For the HH-WQSs, the RSC reflects the percent of total exposure that can be attributed to surface water through water intake (drinking consumption or incidental ingestion) and fish consumption.

The RSC is applied to health-based criteria for noncarcinogenic and nonlinear carcinogenic chemicals. As previously discussed, HH-WQSs for linear carcinogenic compounds are determined with respect to the incremental lifetime risk posed by the presence in water, and are not set with regard to an individual's total risk from all sources of exposure (USEPA 2000b). The application of 0.2 as a default RSC, like the use of a very low incremental excess cancer risk level (1 in 100,000) as compared to background cancer risk, functions to provide conservative accounting for exposure to a pollutant from media other than water and routes other than ingestion. EPA has also considered policy goals in RSC application by stating HH-WQSs should support good health policy by maintaining stringent standards when the environmental concentrations of that pollutant are already low (USEPA 2000b). For example, chloroform is a good example of the use of an RSC for a highly volatile contaminant; since inhalation is a well-known and significant exposure pathway of concern, and detections in surface water are uncommon, limiting surface water exposure with an RSC of 0.2 (80% of chloroform is assumed to come from other sources) is appropriate to protect public health.

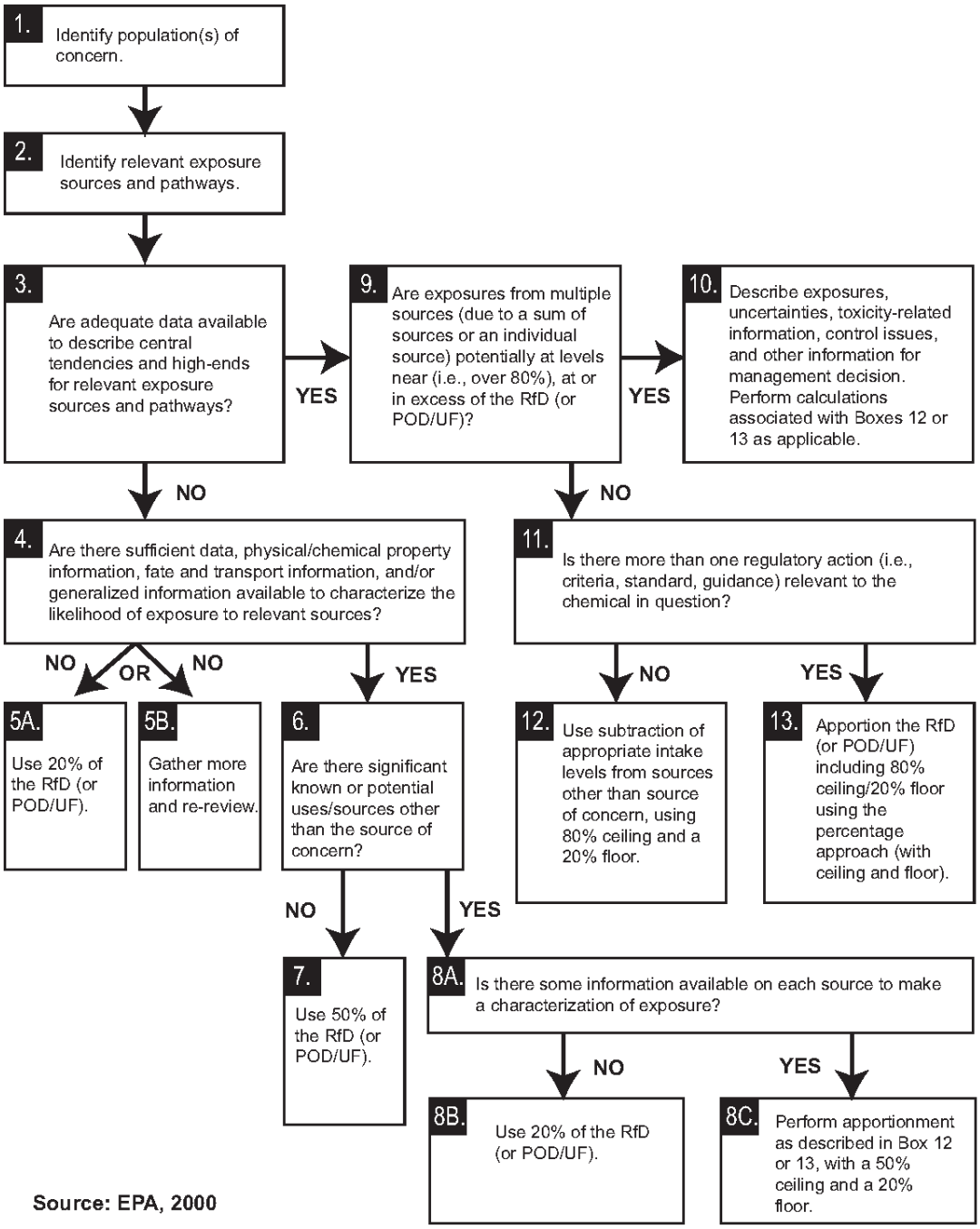
The new RSC approach currently recommended by EPA is called the Exposure Decision Tree, or EDT (Figure 5, USEPA 2000b). This approach has been utilized by MDH in the 2009 HRL rule revisions. The EDT consists of a series of decision points at which the availability and quality of chemical and exposure data are evaluated (MDH 2008a). In general, if adequate data are not available to describe central tendencies and high-ends for relevant exposure sources and pathways, an RSC between 0.2 and 0.5 is chosen based on the relative level of knowledge of

known and potential sources and routes of exposures. When chemical-specific data are sufficient to describe other sources and routes of exposure, a pollutant-specific RSC can be develop, but still has a "ceiling" RSC of 0.8 (USEPA 2000b). The mercury fish-tissue criterion was the first EPA criteria to use a chemical-specific RSC and the subtraction method. The new guidance from EPA for RSC, as described below, will not result in a significant change in the rule, but better describes the approach used when developing RSCs. MPCA's approach for RSCs has historically and will continue to be consistent with MDH's approach.

In Boxes one and two the EDT formulates the problem. The HH-WQS are intended to be protective of the general population, including sensitive subpopulations. Relevant exposure pathways may differ dependent upon the particular contaminant, but may include inhalation, dermal absorption, and food consumption. Box three asks if adequate data are available to calculate exposure from relevant pathways. For most pollutant, MPCA does not expect there to be adequate ambient sampling data to statistically calculate exposure to the general population to develop chemical-specific percentile RSCs or subtraction units. On the other hand, there is generalized information available to characterize the likelihood of exposure to relevant sources, which is asked in Box four. Box six leads to three different outcomes. An RSC of 0.5 is used if there are not additional significant known or potential sources of the contaminant. If data are available to show that there are significant additional sources of the contaminant, but information is insufficient to make a full characterization of exposure, an RSC of 0.2 is used. Lastly, an RSC between 0.2 and 0.5 can be developed if there are significant additional sources and if there is information available to perform an apportionment.

MPCA will most often use an RSC of 0.2 unless, based on professional judgment it is estimated that there are no other significant sources of or routes of exposure to a particular chemical besides drinking water, incidental ingestion, and fish consumption; for those chemicals MPCA would apply an RSC of 0.5. In addition, chemical-specific RSCs may be developed based on the EDT guidance, EPA chemical-specific *Ambient Water Quality Criteria*, or MDH-developed RSCs used in HRLs. This policy use of the RSC ensures that the HH-WQS will be protective of human health even when there are other routes of exposure for a particular contaminant, but allows for less stringent assumptions when multiple sources of exposure are not anticipated. MPCA's approach is consistent with MDH's in the inclusion of other non-ingestions routes of exposure from surface water in the RSC determination. Again using chloroform as an example, the RSC of 0.2 would be used, because a significant route of exposure is inhalation.

Figure 5: Relative Source Contribution (RSC) Exposure decision tree from USEPA 2000b



Source: EPA, 2000

D. Great Lakes Initiative standards in Minn. R. ch. 7052

As introduced in Section II.D., Minn. R. ch. 7052 is Minnesota's promulgation of the GLI for the Lake Superior Basin, as required by an amendment to the Clean Water Act (CWA). MPCA adopted by reference the GLI methods and water quality standards as specified in EPA (1995d) and 40 CFR Part 132, Appendices A through D, with little or no change. In general, the GLI

methods reflected advances in the science of standard development; however, specifically for human health methods, they were fundamentally the same as those in Minn. R. ch. 7050, except for the BAF methods. This section discusses background on the methods in the GLI and Minn. R. ch. 7052 and proposed changes needed to reflect the revised methods.

a. Toxicological and exposure evaluations specific to Minn. R. ch. 7052

As discussed in the earlier sections on *Toxicological* and *Exposure Evaluation*s, when the GLI was adopted in Minn. R. ch. 7052 in 1998, the toxicological methods matched those used to develop statewide HH-WQSs in 1990. The human health algorithms in both rules are designed to protect the beneficial uses of drinking water, fish consumption, and recreation for chronic or lifetime durations. Pollutant-specific toxicological reviews were completed by EPA and published in GLI supporting documents (USEPA 1995b, USEPA 1995c). These reviews provided the RfDs and q1* (or cancer slope factors) used by MPCA to develop human health-based chronic standards (CS) listed in Minn. R. ch. 7052. While EPA approved of two approaches to develop toxicological values: Tier I and Tier II (when data were limited), all the GLI criteria for human health protection had enough data to have Tier I toxicological values. EPA has not used and is not expected to use the Tier II method for any subsequent human health-based AWQC, so MPCA will remove reference to this method as part of the proposed revisions. In addition, MDH does not have a comparable method to develop RfDs or CSFs when data are less than required for development of a HRL or HBV (MDH 2008a). Minimal dataset would be chemical-specific and consider Federal requirements for CWA purposes and adherence to State rule language on acceptable and sufficient data.

Where Minn. R. ch. 7052 does differ from the GLI methods is in the use of the higher recreational sport fish consumption rate and fish lipid percentage used in the BAF methods discussed in the next section. By adopting a fish consumption rate of 0.030 kg/day, MPCA maintained consistency with the rate used in Minn. R. ch. 7050 and because the regional survey data included participants that fish the Great Lakes. In addition, MPCA ensured that a higher percentage of fish consumers are protected from adverse health effects caused by bioaccumulative pollutants. The consistency between the fish consumption rates used in Minn. R. chs. 7050 and 7052, and the protection of a higher percentile of consumers, reflects the commitment by MPCA to maintain and improve the safety of fish consumption in Minnesota, especially since tourism and fisheries recreation are extremely important to this state.

With the revision of human health parameters and algorithms, MPCA plans to increase consistency between the two water quality rules and enhance protection for infants and children; therefore, the adult FCR in Minn. R. ch. 7052 will be supplemented by the use of the FCRs for cancer ADAF child age groups, new MDH drinking water intake rates, and MPCA's incidental water rate, applied consistently in the Lake Superior Basin and statewide HH-WQSs as described fully in Section V.

b. Bioaccumulation factors

The methods adopted in 1990 in Minn. R. ch. 7050 to determine BAFs and BCFs for use in HH-WQS were developed primarily by MPCA as EPA did not have guidance on these methods until the GLI. As part of the GLI, methods were developed to determine BAFs based on a pollutant's

physicochemical properties and hierarchy of available data (USEPA 1995a). While these methods differed from those in Minn. R. ch. 7050 (see Appendix B2), the new revised BAF methods discussed in this guidance as set forth by EPA in 2000 are planned to replace the 1990 methods and are based on and almost identical to the GLI methods already in Minn. R. ch. 7052 (see Section IV.C.g.). Where the use of state or local data have been supported by EPA guidance in place of national defaults, for example, in the use local fish lipid (and consumption data-see above) or water chemistry parameters (e.g., POC and DOC), Minn. R. ch. 7052 will continue to use the Minnesota fish lipid percentages for Lake Superior Class 2A (8.5%) and Class 2A cold water or trout fisheries (6%) for TL3 and TL4. The only revision will be for Class 2B waters, with a new TL3 lipid value of 2% and 1.5% for TL4 (see Appendix B3); to calculate the f_{fd}, the GLI POC and DOC values will also remain those developed in the GLI and applied to BAFs developed for the Lake Superior Basin.

E. Fish tissue human health-based water quality numeric and narrative standards

As discussed earlier, HH-WQSs have their foundation in EPA *Ambient Water Quality Criteria* (AWQC) and for many reasons the criteria have been developed as water concentrations. Water concentrations are most appropriate for applying protection to aquatic organisms from toxic pollutants, implementing WQSs in point source effluent limits, and monitoring for non-bioaccumulative pollutants. However, for pollutants that are highly bioaccumulative, fish tissue is the medium that should be addressed, both for monitoring and HH-WQS application. EPA supports this approach and in 2001 published the first fish tissue residue AWQC for methylmercury. In 2008, MPCA adopted the fish tissue HH-WQSs for mercury in Minn. R. ch. 7050. Fish tissue HH-WQSs for highly bioaccumulative pollutants–which MPCA is defining by using the BAF threshold from the GLI BCC definition –provide a more accurate way to assess and to determine that human health protection for fish consumers is being met.

The inclusion of fish tissue HH-WQSs also fits with the overarching narrative goals for Class 2 WQSs, "nor shall there be any significant increase in harmful pesticide or other residues in the waters, sediments, and aquatic flora and fauna." In 2003 MPCA adopted a more defined narrative standard to implement this goal in Minn. R. 7050.0150, subp. 7 that incorporates the MDH Fish Consumption Advice (FCA) (see Section III.B.). In short, if a waterbody has advice that limits fish consumption to levels more restrictive than one meal per week for any population, the goals of HH-WQSs are not being met and that water is considered impaired. With the addition of the mercury fish tissue HH-WQS, that standard is now the basis for waters being listed as impaired for mercury (it is more restrictive than current FCA). More discussion on implementing HH-WQSs is in Section VI.

While the reference to FCA supports the goals of HH-WQS, there are gaps in this approach when relied on for assessing if WQSs are being met and for implementing pollution controls. One difference stems for the vital role of FCA in balancing the risk and benefits of fish consumption (MDH 2009). This goal is accomplished through consideration of the nutritional value of eating fish, while also advising consumption of fish containing lower amounts of

mercury. As a basis for pollutant-specific FCA thresholds, this goal may mean use of different toxicological values or exposure parameters.

In addition, cooperation with other states in the region to maintain consistency in advice, which can result in different fish consumption rates or toxicological values than those used in HH-WQSs. FCA also is specific to the pollutant with the most restrictive advice; whereas, the use of fish tissue based CS would address any pollutant that had concentrations exceeding its numeric value given minimum data requirements (MPCA 2012). For example, when the concentrations of mercury in fish results in more restrictive advice than for PCBs and on occasion other legacy pollutants, MPCA's reliance solely on mercury advice for comparison to standards wouldn't necessarily addressed all HH-WQSs not being met.

Based on the GLI, a toxic pollutant that, in short, *accumulates in aquatic organisms by a human health BAF greater than 1,000, after considering metabolism and other physiochemical properties that might enhance or inhibit bioaccumulation*, is considered a BCC. MPCA will use this threshold BAF to identify toxic pollutants to add fish tissue HH-WQSs to supplement the water human health-based CS. The use of a final BAF > 1,000 also fits the general definition of a bioaccumulative pollutants and reflects the physicochemical properties of pollutants most likely to biomagnify: log octanol-water partitioning coefficients (log K_{ow}) greater than 4 and be reliably measured in fish tissue. In addition, when BAFs are 1,000 or greater, the drinking water or incidental ingestion routes of exposure are negligible and the HH-WQS algorithms need only reflect the fish consumption pathway.

As MPCA primarily relies on MDH toxicological reviews or their methods as incorporated in the revised methods for future development of new or revised HH-WQSs, the addition of fish tissue standards will be phased in as new reviews are available from MDH or from MPCA based on the revised methods. Prior to new or revised water and fish tissue HH-WQSs being complete for all pollutants with final BAFs greater than 1,000 (defined BCC threshold) in Minn. R. chs. 7050 and 7052 using the revised methods, MPCA would consult with MDH on the use of more current toxicological values if needed to evaluate those pollutants detected in fish tissue. The basis for this consultation and others with MDH is described in Section V.C. If there are significantly different toxicological values than use currently in place as the basis for listed water-based CS, MPCA would take the steps to gain EPA approval of a site-specific modified standard for that CS and subsequent application of the toxicological values in the fish tissue-based algorithms.

The parameters used for the new algorithms are a subset of those used for water HH-WQSs. The big difference being that besides only using FCRs, no BAFs are required. Development of a final fish tissue HH-WQSs is based on the use of two specific algorithms and implementation of the most stringent value calculated from them. There is one algorithm for addressing noncancer or nonlinear carcinogenic effects. If the BCC is also a linear carcinogen, a second algorithm is used depending on if the CSF has a lifetime or no adjustment factors using the FCR_{adult} or is applied with a ADAFs and duration specific FCRs. The details are found in Section V.

V. Revised Parameters and Algorithms for Human Health-based Water Quality Standards

A. Proposed parameters for HH-WQS by toxicological profile and use classification

a. Introduction

The detailed sections, *Toxicological Evaluations* and *Exposure Evaluations*, provide the foundation for the proposed revisions to the algorithms used to develop human health-based water quality standards (HH-WQSs), specifically chronic standards (CSs) or site-specific Chronic Criteria (CC). As seen in Table 6, many of the parameters (both toxicological and exposure related) used in the algorithms have substantially changed since 1990 with one key principle underscoring much of the revisions, improving protection to infants and children that may contact surface water pollutants through drinking water, fish consumption, or recreational activities.

The following sections describe the basis for the revised parameters and how they will be applied in the revised algorithms. Of particular relevance, are new algorithms that will set HH-WQSs in fish tissue for Bioaccumulative Chemicals of Concern (BCC); this addition will assist MPCA in evaluation of those pollutants to determine if WQSs are being met in our water resources. The discussion also includes how the revisions will include better methods to address risks from multiple chemicals, including assessing chemical mixtures and degradates.

b. Revised toxicological parameters

The goal of HH-WQSs is to be consistent with and as protective as MDH's HRL program and toxicological evaluation methods. When revising existing or adopting new HH-WQSs, the same RfDs and CSFs and drinking water intake rates used to calculate HRLs, HBVs or other health-based guidance (drinking water values developed based on HRL methods, but not promulgated into rule) will be used when available. RfDs and CSFs obtained from EPA Integrated Risk Information Service (IRIS), HBVs, or other sources will be used after evaluation using the new HH-WQS methods. The procedures used by MDH staff to evaluate RfDs, CSFs, and other human health data for the development of HRLs are described in MDH's 2008 SONAR and Minn. R. 4717.7810 to 4717.7900 are the basis for the revised HH-WQS methods and are incorporated into Minn. R. ch. 7050. Previously adopted 1993/94 HRLs listed in Minn. R. 4717.7500 may have used the same RfDs and CSFs as many existing HH-WQSs; however, in future applications, those toxicological values will not be the basis for new or revised HH-WQS until reviewed again by using HH-WQS updated methods.

Under *Toxicological Parameters-Noncancer*, the two biggest changes are in the development and use of duration-specific RfDs that include less-than-chronic durations and listing of Health Endpoints for multiple chemical standards (see Section IV.B.b. *Noncancer Evaluations*). HH-WQSs are built on chronic or lifetime protection; however, adverse health effects can occur

from shorter duration exposures. EPA's IRIS and state agencies previously approached chronic standards through the almost exclusive use of RfDs based on animal studies and in some cases human epidemiological studies addressing chronic exposure (defined as greater than 10% of an organisms' lifespan). Toxicological data were available and are now more defined in their application for assessing health effects that occur after much shorter durations. MDH has captured these effects by developing multiduration RfDs for acute, short-term, subchronic, and chronic durations. As described in the *Revised Algorithm* Sections V.B.b, and c., HH-WQSs will continue to primarily be based on chronic RfDs, but will incorporate acute, short-term, or subchronic RfDs on a pollutant-specific basis primarily to ensure that the final CS is protective of developmental susceptibility (considering both increased sensitivity for effects and higher exposures).

New to HH-WQSs is the listing of the type of health effect by organ or system, referred to as a Health Endpoint, associated with a pollutant's RfDs. MDH in the HRL rule has employed Health Endpoints as a way to address noncancer risk at a site where multiple pollutants are found in groundwater. The approach centers on limiting the risk of health effects from mixtures of pollutants that act similarly or can affect development. MPCA will include Health Endpoints for the RfD used to set final HH-WQSs to enhance human health protection for surface water users.

While many of the practices and outcomes of cancer assessments have changed since 1990 (see Section IV.B.c. *Cancer Evaluation*), the expanded understanding on the mode of action (MOA) of carcinogenesis and carcinogens has been one of the most significant. The elucidation of MOA information has led to many carcinogens now being characterized more accurately as nonlinear carcinogens from the default linear classification. Many carcinogens will still retain a linear descriptor, assuming that any exposure can lead to cancer, and application of a CSF in HH-WQSs; however, pollutants that have enough MOA data to elucidate the key biological processes that occur and demonstrate there is a threshold below which cancer outcomes are unlikely are evaluated under the noncancer methods. These pollutants will be labeled as (NLC), for nonlinear carcinogens from the previous generic (C) for a carcinogen currently in Minn. R. chs. 7050 and 7052.

Another critical area of new study reflected in the *Toxicological Parameters-Cancer* (Section V.B.c.), is the use of adjustment factors to reflect the higher potency of carcinogens when exposure occurs in a developmental life stage. MPCA has added new parameters to reflect this higher cancer risk. The default approach will be using the ADAF developed by EPA (2005b) and incorporated and more broadly applied by MDH (2008a). When pollutant-specific adjustments are available, MPCA will apply those. The ADAF parameters or a pollutant-specific lifetime Adjustment Factor ($AF_{lifetime}$) will be multiplied by the CSF to reflect the higher cancer potency for the appropriate duration.

c. Revised exposure parameters

The *Exposure Characterization Parameters* have also been reviewed, with important revisions being planned for DWIRs, FCRs, and IWR differences in exposure rates during early life stages, and BAF methods. The intake rates are now all expressed on a per body weight basis to more easily show exposure differences by media and compare to toxicological values (RfDs and CSFs).

The DWIRs[39], as referenced in the HH-WQSs algorithms, will be MDH's drinking water intake rates: either the default rates listed by duration or as developed when pollutant-specific data are available. The recently adopted MDH drinking water intake rates (IRs) are applied on a duration basis and reflect the highest time-weighted exposure rates from surveys of "consumer only" community water use over a person's lifespan. MDH has also moved to using 95th percentile rates to ensure more protective standards for the general population, particularly bottle-fed infants that are more likely than other subpopulations to obtain water from a single household source (MDH 2008a). Depending on the age ranges reviewed, the previous rate of 2 L for a 70 kg adult equates to less than a 50th percentile rate for infants up to 90th percentile rate for most adults. MPCA is incorporating these new MDH IR rates into HH-WQS as DWIRs.

MPCA also examined newer data, including use of higher percentiles, and approaches for improving the IWR. The proposed IWR incorporates newly recommended intake rates for children when swimming from EPA's *Exposure Factors Handbook* (2011), EPA RAGs, and MDH and MPCA site assessments estimates of Reasonable Maximum Exposure for recreational swimming patterns in Minnesota. The IWR also is based specifically on the ages and minimum chronic duration to set this exposure parameter to best reflect children recreating in surface waters most likely to be accidently ingesting water while swimming, ages 1 to 9.

As discussed previously, accounting for exposure to a pollutant in fish is a key aspect of setting HH-WQSs. All rivers and lakes in Minnesota are protected for fish consumption. Revised parameters cover a few different aspects of ensuring pollutant concentrations in fish do not exceed safe levels: amount of fish eaten on a per body weight basis (FCR) and potential for bioaccumulation in fish (based on trophic level and new bioaccumulation factors methods). New FCRs are proposed for use to more accurately account for children's intake of freshwater fish when applying the cancer ADAFs. The revised parameters also include updates based on extensive EPA guidance on BAF methods, including distinguishing fish consumption and BAFs by trophic level (TL). The inclusion of TLs in BAF calculations uses available data to better address the fact that fish differ in their tissue pollutant concentrations. And while not readily apparent in parameters table, the improved BAF methods will mean consistency between the state's two water quality rules and increase accuracy and options for developing BAFs by pollutant type and site characteristics.

The final *Exposure Parameter* is the RSC factor; while the RSC retains its previous definition, the new guidance provided by EPA (2000b) gives a clearer framework, Exposure Decision Tree, and more default values for use in improving how this factor is applied.

In addition, guidance is provided as to scenarios when the default intake rates could be replaced with alternate rates. Pollutant-specific information and additional survey data for use in site-specific application are options for refinement. In addition, a key strength of NHANES involves the inclusion of biomonitoring data along with survey information to provide more accurate estimates of human exposure to environmental pollutants beyond just the use of defaults. NHANES is collecting biomonitoring data for a long list of pollutants that have been

[39] MDH (2008 and Minn. R. ch. 4717) uses the acronym *IR* for the drinking water intake rates; MPCA will use DWIR to distinguish MDH drinking water intake rates from the other *intake rates* used for incidental water ingestion and fish consumption in HH-WQSs.

and will be used to set pollutant-specific exposure parameters in risk assessments for pollutants such as mercury (Mahaffey et al. 2004). Use of biomonitoring data, modeling of internal doses, and survey information on exposure sources and patterns offer more accurate and meaningful estimates of exposure than those based solely on consumption surveys for use in setting protective standards. Biomonitoring and epidemiological data were the basis behind EPA's *Water Quality Criterion for the Protection of Human Health: Mercury* (2001) (see Section II.E.).

d. Table of proposed parameters for use in HH-WQS algorithms

Table 6: Revised parameters for HH-WQSs

Topic area	Parameter	Abbreviation	Value and units	Unit description
Toxicological Characterization-Noncancer (Systemic Toxicants and Nonlinear Carcinogens)	Duration-specific Reference Doses (MDH): More accurately characterize health effects for a given duration of exposure.	$RfD_{acute.}$ $RfD_{short\text{-}term}$ $RfD_{subchronic}$ $RfD_{chronic}$	Pollutant specific value mg/kg-d	Milligram chemical /kilogram body weight-day
	Health Endpoints (MDH): -General description of a toxic effect used to group chemicals (organ or system target of critical toxic effects, e.g. hepatic, immune, kidney, etc.) -Also used to address range of Developmental Effects	"Endpoints" By RfD and Chemical-specific	No Units	
Toxicological Characterization-Carcinogens (for Nonlinear Carcinogens, refer back to *Noncancer Characterization Parameters*)	Cancer Potency Slope Factor for Linear Carcinogens -No changes, except in Acronym use to match MDH (replace q1*)	CSF	Pollutant specific value $(mg/kg\text{-}d)^{-1}$	One over milligram chemical/ kilogram bw-day
	Age-Dependent Adjustment Factors (MDH) -EPA Early-Life Cancer Adjustment Factor, but applied based on MDH protocol (HRL rule). -Adjustment Factor (lifetime)	$ADAF_{<2}$ $ADAF_{2\ to\ <16}$ $ADAF_{16\ to\ 70}$ $AF_{Lifetime}$	10 3 1 Chemical-specific	No Units
	Cancer Risk level -No Changes	CR	10^{-5}	1 in 100,000 excess cancer risk

Topic area	Parameter	Abbreviation	Value and units	Unit description
Exposure Characterization	Drinking Water Intake Rate (MDH)- -Replace Adult Rate - Rates include time-weighted intakes and age-adjusted BWs -Current rates are MDH defaults based on the 95th percentile from survey data applied in 2008. -Can also be chemical-specific.	$DWIR_{acute}$ $DWIR_{short\text{-}term}$ $DWIR_{subchronic}$ $DWIR_{chronic}$ $DWIR_{0-<2}$ $DWIR_{2\ to\ <16}$ $DWIR_{16\ to\ 70}$ $DWIR_{lifetime}$	0.289 L/kg-d 0.289 L/kg-d 0.077 L/kg-d 0.043 L/kg-d 0.137 L/kg-d 0.047 L/kg-d 0.039 L/kg-d 0.043 L/kg-d Or Pollutant Specific	Liters/ kilogram body weight-day
	Incidental Water intake Rate (Swimming Exposure) (MPCA) -Revising based on Minnesota specific swimming activity patterns and data on children from 2008 EPA Exposure Factors Handbook	IWR	Revised: 0.0013 L/kg-d (Existing: 0.00014 L/kg-d)	Liters/kilogram body weight-day
	Fish Consumption Rate (MPCA)-freshwater fish intake rate-adult -New Child Rates -Rates include age-adjusted BW	FCR_{adult} $FCR_{child\ (1-5)}$ $FCR_{0\ to\ <2}$ $FCR_{2\ to\ <16}$	0.43 g/kg-d (30 g of fish/70 kg BW) 0.86 g/kg-d 0.86 g/kg-d 0.55 g/kg-d	Grams fish /kilograms body weight-day (In algorithms: x 10^{-3} to convert to kg (fish)/kg(bw)-d)
	Bioaccumulation Factor (MPCA)- *State*-BAF -Revising Methods to calculate final BAFs to reflect GLI; -Adding Trophic Levels (TLs) to statewide rule based on GLI - 76% to TL4 and 24% to TL3; -New TL lipid values for Class 2B	BAF_{TL3} (24%) BAF_{TL4} (76%)	Pollutant- and Use Class specific value L/kg	Liters/kilogram fish tissue
	Relative Source Contribution factor -Based on EPA 2000 RSC Exposure Decision Tree	RSC	Pollutant-specific value between 0.2 and 0.8 or defaults of 0.2 or 0.5	Percent (no units) or Subtraction method (mg/kg-d)
	Body Weight-adult -Include with Intake Rates	—	—	—

B. Proposed algorithms for HH-WQS by toxicological profile and use classification

a. Revised algorithms-introduction

As described in this section, HH-WQSs (MPCA) and CWA 304(a) AWQC (EPA) have always been based in providing long-term or chronic protection from contaminants in surface water (Section III.C.). MPCA is maintaining HH-WQSs as "Chronic Standards", but is strengthening this approach by explicitly considering developmental susceptibility in the revised algorithms. As recognized by MDH, to ensure lifetime protection, health protective standards must also account for shorter windows of vulnerability to toxic effects and higher exposure rates to ensure full lifetime protection goals are met.

The underlying algorithm (equations and steps) for calculating any HH-WQS is:

$$Class\ 2\ HH-WQS, Chronic\ (CS/CC) = Toxicological\ Value\ /\ Exposure$$

where, the *Toxicological Value* is either based on protection from noncancer (or nonlinear carcinogenic) adverse effects using a RfD or linear carcinogenic potency expressed by a CSF, and now, early-life potency adjustment factors (ADAF or $AF_{Lifetime}$). *Exposure* represents the intake rates for the specific beneficial uses surface waters are classified for: drinking water source, fish consumption, and incidental water for recreational activities. Specific algorithms are presented below that build off of this basic equation and address the details for application.

As noted with each specific algorithm, the CS (or site-specific CC) developed as a water concentration will be distinguished by the exposure routes used for each use classification. For CSs applicable for Class 2A and 2Bd waters that protect drinking water sources, recreation (higher DWIRs sufficiently address IWR), and fish consumption, the acronym used is CS_{dfr}. For Class 2B (2C and 2D) where drinking water use is not included and the exposure is based on recreational exposure (IWR only) and fish consumption, the acronym is CS_{fr} (previously referred to as CS_f). When a CS is developed for a pollutant with acute, short-term, or subchronic developmental or less-than-chronic toxicity and higher early-life exposure rates, the acronym will be CS_{dev}. If more than one CS can be developed for a pollutant (e.g., for a pollutant with a chronic RfD and a less-than-chronic toxicity-based RfD or a chronic RfD and linear carcinogen), the more stringent CS by use classification (CS_{dfr}/CS_{dev} or CS_{fr}/CS_{dev}) will be listed as the final applicable HH-WQS (see Section VI.B.)[40].

When fish tissue CSs (or CCs) are developed for pollutants meeting the BAF threshold in the BCC definition, the standard will be described as CS_{ft}; the negligible exposure from water routes means these algorithms do not include DWIRs or IWR (see Section IV.E.). In addition, as CS_{ft} does not involve BAF calculations, no distinctions are made on the basis of fish characteristics: TL or lipid differences. The following sections include three algorithms for CS_{ft}: noncancer (nonlinear carcinogen), linear carcinogen with $AF_{lifetime}$ or no adjustment factors, and linear carcinogen with ADAFs. The most stringent CS_{ft} will be the final HH-WQS listed in rule.

[40] CSs will be listed with two significant figures for consistency with EPA's AWQC (USEPA 200b). MPCA will round down the final value to ensure final CSs are below and not higher than the calculated values.

b. Noncancer or nonlinear carcinogen algorithms

Below are the general algorithms for developing the HH-WQSs (CSs or CC) for pollutants characterized for noncancer adverse effects. The basis for the algorithm is to ensure that exposure from surface water and other sources will not exceed the RfD, the dose derived to be protective of adverse systemic toxic effects for a specific duration of exposure. When the pollutant has also demonstrated cancer potential through a nonlinear or threshold mode of action, the RfD is also used to set the protective dose; therefore, CSs for nonlinear carcinogens (NLCs) will also be derived using the same algorithms. What follows are three specific algorithms that will be listed in Minn. R. chs. 7050 and 7052, including the details on the application of $RfD_{duration}$ and exposure by Use Classification for both water and fish tissue CSs.

$$CS\ (Water\ Column)\ \ in\ \mu g/L = RfD_{Duration}\ (mg/kg-d)\ x\ \frac{Relative\ Source\ Contribution\ (no\ units)}{Water\ Exposure\ (L/kg) + Fish\ Exposure\ (L/kg)}\ x\ 1000\ \frac{\mu g}{mg}$$

$$CS\ (Fish\ Tissue) in\ mg/kg\ (ppm) = \frac{RfD_{Duration}\ (mg/kg-d)\ x\ RSC\ (no\ units)\ or - RSC\ (mg/kg-d)}{Fish\ Consumption\ Rate\ (kg/kg-d)}$$

The first two noncancer algorithms are always examined when developing a CS for a pollutant. The foundation for these algorithms is the same as the 1990 noncancer algorithms, because of the consistency of these methods in the use of chronic duration RfDs and intake rates. The difference is that the intake rates are now better defined to more accurately reflect long-term drinking water and incidental water rates. Improvements to the BAF and RSC methods are also reflected in their parameter descriptions. The BAFs for all algorithms will now be defined by two Trophic Level (3 and 4) values that are applied proportionally to reflect recreational Great Lakes regional fish consumption patterns. The only difference in the algorithms is the use of either the chronic DWIR or IWR to reflect the difference in use classifications between Class 2A or 2Bd and Class 2B (2C and 2D).

Revised DW+FC Noncancer Algorithm 1

Beneficial Use Classification
Class 2A or 2Bd: Drinking Water source, Fish Consumption, and Recreation, CS_{dfr} (µg/L)

$$RfD_{chronic}\ (mg/kg-d)\ x\ \frac{RSC\ (no\ units)\ x\ 1000\ \frac{\mu g}{mg}}{\{DWIR_{chronic}\ (L/kg-d)\ +\ FCR_{adult}\ (kg/kg-d)\ [(0.24\ x\ BAF_{TL3}\ (L/kg))\ +\ (0.76\ x\ BAF_{TL4}(L/kg))]\}}$$

$RfD_{chronic}$ And list Health Endpoints	Reference Dose (RfD) designed for chronic durations (> 10% of lifetime); comparable to 1990 RfD definition.	Pollutant-specific in mg/kg-d (sources: MPCA, MDH and EPA)
Water Exposure	Drinking Water Intake Rate (DWIR) = MDH $IR_{chronic}$ (or pollutant-specific)	0.043 L/kg-d (default)
Fish Exposure	Fish Consumption Rate (FCR): adult	0.00043 kg/kg-d (0.43 g/kg-d) (default)
	Bioaccumulation Factors (BAFs): *State*-BAFs and Lake Superior-BAFs proportioned by Trophic Level (TL) 4 at 76% and TL 3 at 24% & Lipid content 8.5% Lake Superior; 6% for Class 2A and 1.5% TL4 and 2% TL3 for Class 2Bd	Pollutant-specific in L/kg (sources: MPCA and EPA)
Relative Source Contribution (RSC)	Based on EPA 2000 Exposure Decision Tree: Accounts for exposures other than ingestion of water and fish	Pollutant-specific or more often: 0.2 or 0.5 (defaults) (sources: MPCA, EPA and MDH)

Revised IW+FC Noncancer Algorithm 1

Beneficial Use Classification
Class 2B: Fish Consumption and Recreation (Incidental Water exposure), CS_{fr} (µg/L)

$$RfD_{chronic}\ (mg/kg-d)\ x \frac{RSC\ (no\ units)\ x\ 1000\ \frac{\mu g}{mg}}{\{IWR\ (L/kg-d)\ +\ FCR_{adult}\ (kg/kg-d)\ [(0.24\ x\ BAF_{TL3}\ (L/kg))\ +\ (0.76\ x\ BAF_{TL4}(L/kg))]\}}$$

$RfD_{chronic}$ And list Health Endpoints	Reference Dose (RfD) designed for chronic durations (> 10% of lifetime); comparable to 1990 RfD definition.	Pollutant-specific in mg/kg-d (sources: MPCA, MDH and EPA)
Water Exposure	Incidental Water Intake Rate (IWR)	0.0013 L/kg-d
Fish Exposure	Fish Consumption Rate (FCR): adult	0.00043 kg/kg-d (0.43 g/kg-d) (default)
	Bioaccumulation Factors (BAFs): State BAFs proportioned by Trophic Level (TL) 4 at 76% and TL 3 at 24% & Lipid content of 1.5% TL4 and 2% TL3 for Class 2B	Pollutant-specific in L/kg (sources: MPCA and EPA)
Relative Source Contribution (RSC)	Based on EPA 2000 Exposure Decision Tree: Accounts for exposures other than ingestion of water and fish	Pollutant-specific or more often: 0.2 or 0.5 (defaults) (sources: MPCA, EPA, and MDH)

New Fish Tissue Noncancer Algorithm 1

Beneficial Use Classification
Class 2A, 2Bd, 2B: Fish Consumption, pollutant with a final BAF > 1,000, CS_{ft} (mg/kg)

$$= \frac{RfD_{Chronic}\ (mg/kg-d)\ x\ RSC\ (no\ units)\ or - RSC\ (mg/kg-d)}{FCR_{Adult}\ (kg/kg-d)}$$

$RfD_{chronic}$ And list Health Endpoints	Reference Dose (RfD) designed for chronic durations (> 10% of lifetime); comparable to 1990 RfD definition.	Pollutant-specific in mg/kg-d (sources: MPCA, MDH and EPA)
Fish Exposure	Fish Consumption Rate (FCR): adult	0.00043 kg/kg-d (0.43 g/kg-d) (default)
Relative Source Contribution (RSC)	Based on EPA 2000 Exposure Decision Tree: Accounts for exposures other than ingestion of fish; as with other RSC applications, can include percentage or subtraction approach. Subtraction was used for the 2008 fish tissue- based mercury CS_{ft}	Pollutant-specific or more often: 0.2 or 0.5 (defaults) (sources: MPCA and EPA)

c. Supplemental algorithms for developmental susceptibility

Described in detail in *Defining Developmental Protection* (Section IV. A), the profile of a developmental toxicant is such that the prenatal and neonatal life stages may be uniquely susceptible to toxic insults from environmental pollutants with these effects evaluated and addressed by MDH's new less-than-chronic RfDs, IRs, and final duration-specific HRLs or other health-based values (MDH 2008a). As reviewed with MDH, there are specific aspects to developmental toxicity profiles and less-than-chronic durations that require different considerations when applying those RfDs to the drinking water as compared to the fish and incidental water pathways:

- While these windows of time may be susceptible, the dose at which developmental effects occur is not always the lowest when considering any adverse effect observed for that same shorter duration exposure available for any life stage.
- For many pollutants, given the nature of considering long-term, repeated exposure, most chronic adverse effects actually occur at lower doses; in fact, for all the chemicals MDH reviewed for the 2009 HRL rule revision, none had less-than-chronic RfDs for any *Health Endpoint* (including Developmental) that was more stringent than the chronic RfD (Minn. R. 4717.7860).
- When considering pollutant exposure through drinking water consumption, the neonatal life stage has the highest intake rate for the subpopulation of infants consuming formula reconstituted using household drinking water. MDH has used the neonatal life stage from birth to 3 months as the basis for setting HRLs, because not only does this age group have the potential for the highest drinking water exposure of any subpopulation or age group, they are also more sensitive to toxic effects given lack of mature metabolic systems and rapid development. This life stage is also considered relevant to *Developmental Health Endpoints* based prenatal toxicity studies.
- Direct exposure to pollutants from fish and incidental water consumption is not expected to begin until older ages; and while ages one through five may have higher exposure than all older age groups, RfDs derived from prenatal toxicity studies (*Developmental Health Endpoints*) wouldn't be applicable to age groups outside the prenatal and neonatal windows.
- If pollutant-specific toxicological data show the window of highest developmental sensitivity would extend beyond or apply to this later developmental life stage, MPCA would evaluate the use of a children's FCR; likewise, MDH would develop an alternate drinking water intake rate for different durations or life stages calculated using the methods described in the *Drinking Water Consumption* section (95th percentile, time-weighted average).

In summary, pollutant exposures during the prenatal and neonatal life stages are different and complex. For drinking water uses, MDH had examined indirect exposure rates based on pregnant and lactating women and direct neonatal rates for infants fed reconstituted formula with tap water (MDH 2008a). MDH determined that the most protective (and highest for any life stage of this duration) and appropriate drinking water intake rate to apply with an acute or short-term developmental RfD was the neonatal rate with a duration from birth to age 3 months ($IR_{acute/short-term}$ = 0.289 L/kg-d). MPCA considers the most relevant subpopulation for addressing fish intake for developmental toxicants may often be women who are or plan to be pregnant; as previously discussed the need to consider this indirect exposure route and an alternate FCR would be addressed on a pollutant-specific basis and development of an alternate FCR if available and reliable data provide evidence that the FCR_{adult} is not representative of this subpopulation of fish consumers. Likewise, if data are available demonstrating a pollutant targets organs or systems (through a known MOA) relevant to later ages when children start to eat fish, then use of a children's FCR may be appropriate to apply with acute to subchronic RfDs. In those scenarios,

incidental water exposure would also be relevant for inclusion in the developmental algorithm. MPCA would consider these alternate pollutant-specific scenarios based on the window of sensitivity and available data on intake rates in consultation with EPA and MDH.

For most pollutants where later developmental processes are not a target, drinking water exposure only for less-than-chronic durations will be included in a new algorithm, CS_{dev}, to ensure that the final CS_{dfr} or CS_{fr} evaluated first using the algorithms for chronic toxicity and effects is protective for shorter durations. This scenario is known to be relevant for currently reviewed HRL chemicals, and so inclusion of this new algorithm (New *DW Supplemental Developmental Algorithm*) in rule is warranted. This new algorithm helps implement the newer goals of risk assessments science and policy to better ensure protection to infants and children.

New DW Supplemental Dev. Algorithm 1

Beneficial Use Classifications

Class 2A or 2Bd: Drinking Water source, CS_{dev} (µg/L)

$$= RfD_{duration\ (acute\ to\ subchronic)}\ (mg/kg-d)\ x\ \frac{RSC\ (no\ units)}{DWIR_{duration\ (acute\ to\ subchronic)}(L/kg-d)}\ x\ 1000\ \frac{\mu g}{mg}$$

$RfD_{duration(dev)}$ = $RfD_{acute/short\text{-}term,\ or\ subchronic}$ *And list Health Endpoints*	Reference Dose (RfD) based on less than chronic duration	Pollutant-specific in mg/kg-d (sources: MPCA, MDH and EPA)
Water Exposure	Drinking Water Intake Rate (DWIR) = MDH IR_{acute} MDH $IR_{short\text{-}term}$ MDH $IR_{subchronic}$ (or pollutant-specific)	 0.289 L/kg-d (default) 0.289 L/kg-d (default) 0.077 L/kg-d (default)
Relative Source Contribution (RSC)	Based on EPA 2000 Exposure Decision Tree: Accounts for exposures other than ingestion of water (Follows MDH)	Pollutant-specific or more often: 0.2 or 0.5 (defaults) (source: MDH)

d. Linear carcinogen algorithms with lifetime adjustment factor

Like the noncancer methods, new parameters and algorithms are needed for HH-WQSs for linear carcinogens. New changes stem from more refined and accurate use of exposure data and adjustments to cancer potency slope factors (CSFs) to account for life stage and lifetime susceptibility differences. The basic algorithms are below.

$$CS\ (Water\ Column) in\ \mu g/L = \frac{CR\ (no\ units)}{SF\ (mg/kg-d)^{-1}\ x\ ADAF/AF\ (no\ units)}\ x\ \frac{1}{Water\ Exposure\ (L/kg) + Fish\ Exposure\ (L/kg)}\ x\ 1000\ \frac{\mu g}{mg}$$

$$CS\ (Fish\ Tissue)\ in\ mg/kg\ (ppm) = \frac{CR\ (no\ units)}{SF\ (mg/kg-d)^{-1}\ x\ ADAF/AF\ (no\ units)}\ x\ \frac{1}{Fish\ Consumption\ Rate\ (kg/kg-d)}$$

While the methods for developing CSFs have improved since 1990, their use in the cancer algorithms is still the same: applied on the basis of a set level of "incremental excess Cancer Risk". The risk level used by MPCA and MDH in setting standards assumes an upper bound limit of one additional case of cancer in a population of 100,000 people (or CR = 1 x 10^{-5}), where the population has been exposed daily at the standard concentration for 70 years. What is new, as reflected in the following algorithms, is that the cancer potency for linear carcinogens is now reviewed by MDH to put into practice EPA guidance to address full lifetime risk (USEPA 2005a, USEPA 2005b, MDH 2008a). MPCA has incorporated this same approach into the revised methods. The cancer characterization will include one of two options to address this risk.

1) Data are available to estimate a higher lifetime potency associated with exposure in early life stages; this additional potency, which may equal *one* if there is no early-life susceptibility, is included as the $AF_{lifetime}$ with the CSF (covered in this section); or
2) Pollutant-specific data are not available to assess early-life susceptibility and the default ADAFs will be applied (see the following section).

The basic algorithm for linear carcinogens is expanded to show the application of the $AF_{lifetime}$ in conjunction with the CSF. The algorithm includes the new MDH drinking water $IR_{lifetime}$ or MPCA IWR and MPCA FCR for adults as applied previously, except now with better estimates of bioaccumulation as is also the case for the noncancer algorithms. Because the lifetime duration is assumed to be 70 years and is the basis for all parameters, that value is not explicitly listed.

Revised DW+FC Linear Cancer Algorithm 1

Beneficial Use Classification
Class 2A or 2Bd: Drinking Water source, Fish Consumption, and Recreation, CS_{dfr} (µg/L)

$$\frac{CR(1\ x\ 10^{-5})}{CSF(mg/kg-d)^{-1}\ x\ AF_{Lifetime}}\ x\ \frac{1000\ \frac{\mu g}{mg}}{\{DWIR_{Lifetime}(L/kg-d)\ +\ FCR_{Adult}\ (kg/kg-d)\ [(0.24\ x\ BAF_{TL3}\ (L/kg))\ +\ (0.76\ x\ BAF_{TL4}(L/kg))]\}}$$

CSF	Cancer potency Slope Factor (CSF) for linear carcinogens	Pollutant-specific in $(mg/kg\text{-}d)^{-1}$ (sources: MPCA, MDH and EPA)
$AF_{Lifetime}$	The CSF Adjustment Factor-Lifetime ($AF_{Lifetime}$) is used when data are available to examine the additional excess risk associated with pollutant if exposed in early life. The $AF_{Lifetime}$ would replace the default ADAF approach and could equal one if no additional risk is expected.	Pollutant-specific (sources: MPCA, MDH and EPA)
CR	Incremental excess Cancer Risk Level	1 in 100,000 (1 x 10^{-5})
Water Exposure	Drinking water Intake Rate (DWIR) = MDH $IR_{Lifetime}$ (or pollutant-specific)	0.043 L/kg-d (default)
Fish Exposure	Fish Consumption Rate (FCR) is adult rate	0.00043 kg/kg-d (0.43 g/kg-d) (default)
	Bioaccumulation Factors (BAFs): State BAFs apportioned by Trophic Level (TL) 4 at 76% and TL 3 at 24% & Lipid content 6% for Class 2A and 1.5% TL4 and 2% TL3 for Class 2Bd	Pollutant-specific in L/kg (sources: MPCA and EPA)

Revised IW + FC Linear Cancer Algorithm 1

Beneficial Use Classification
Class 2B: Fish Consumption and Recreation (Incidental Water exposure), CS_{fr} (µg/L)

$$\frac{CR(1\ x\ 10^{-5})}{CSF(mg/kg-d)^{-1}\ x\ AF_{Lifetime}}\ x\ \frac{1000\ \frac{\mu g}{mg}}{\{IWR\ \ (L/kg-d)\ +\ FCR_{Adult}\ (kg/kg-d)\ [(0.24\ x\ BAF_{TL3}\ (L/kg))\ +\ (0.76\ x\ BAF_{TL4}(L/kg))]\}}$$

CSF	Cancer potency Slope Factor (CSF) for linear carcinogens	Pollutant-specific in $(mg/kg\text{-}d)^{-1}$ (sources: MPCA, MDH and EPA)
$AF_{Lifetime}$	The CSF Adjustment Factor-Lifetime ($AF_{Lifetime}$) is used when data are available to examine the additional excess risk associated with pollutant if exposed in early life. The $AF_{Lifetime}$ would replace the default ADAF approach and could equal one if no additional risk is expected.	Pollutant-specific (sources: MDH and EPA)
CR	Incremental excess Cancer Risk Level	1 in 100,000 (1×10^{-5})
Water Exposure	Incidental water Intake Rate (IWR)	0.0013 L/kg-d
Fish Exposure	Fish Consumption Rate (FCR) is adult rate	0.00043 kg/kg-d (0.43 g/kg-d) (default)
	Bioaccumulation Factors (BAFs): State BAFs proportioned by Trophic Level (TL) 4 at 76% and TL 3 at 24% & Lipid content 1.5% TL4 and 2% TL3 for Class 2B	Pollutant-specific in L/kg (sources: MPCA and EPA)

New Fish Tissue Linear Cancer Algorithm 1

Beneficial Use Classification
Class 2A, 2Bd, 2B: Fish Consumption, pollutant with a final BAF > 1,000, CS_{ft} (mg/kg)

$$=\ \frac{CR\ (1\ x\ 10^{-5})}{CSF\ (mg/kg-d)^{-1}\ x\ AF_{\ lifetime}(no\ units)}\ x\ \frac{1}{FCR_{Adult}\ (kg/kg-d)}$$

CSF	Cancer potency Slope Factor (CSF) for linear carcinogens	Pollutant-specific in $(mg/kg\text{-}d)^{-1}$ (sources: MPCA, MDH and EPA)
$AF_{Lifetime}$	The CSF Adjustment Factor-Lifetime ($AF_{Lifetime}$) is used when data are available to examine the additional excess risk associated with pollutant if exposed in early life. The $AF_{Lifetime}$ would replace the default ADAF approach and could equal one if no additional risk is expected.	Pollutant-specific (sources: MPCA, MDH and EPA)
CR	Incremental excess Cancer Risk Level	1 in 100,000 (1×10^{-5})
Fish Exposure	Fish Consumption Rate (FCR) is adult rate	0.00043 kg/kg-d (0.43 g/kg-d) (default)

e. Linear carcinogen algorithms with age dependent adjustment factors

The algorithms that incorporate the new Age Dependent Adjustment Factors (ADAFs) also build from the basic equation, but have to address exposure and cancer risk specifically in the age groups defined by EPA in the *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (2005b). For the three age groups—birth up to less than 2 years of age, 2 up to less than 16 years of age, and 16 up to 70 years of age—MDH had determined appropriate drinking water IRs and MPCA FCRs. MPCA is applying the single IWR to all age groups. The duration (D) in years for each age group is included in the algorithm and divided by the total lifetime averaging duration of 70 years.

Beneficial Use Classification

Class 2A or 2Bd: Drinking Water source, Fish Consumption, and Recreation, CS_{dfr} (µg/L)

$$\frac{CR\,(1 \times 10^{-5}) \times 1000}{\left(\begin{array}{l}\{CSF \times ADAF_{<2} \times D_{<2} \times [DWIR_{<2} + FCR_{<2} \times (0.24BAF_{TL3} + 0.76BAF_{TL4})]\} + \\ CSF \times ADAF_{2\,to\,<16} \times D_{2to<16} \times [DWIR_{2\,to<16} + FCR_{2\,to<16} \times (0.24BAF_{TL3} + 0.76BAF_{TL4})] + \\ CSF \times ADAF_{16to70} \times D_{16to70} \times [DWIR_{16to70} + FCR_{Adult} \times (0.24BAF_{TL3} + 0.76BAF_{TL4})]\end{array}\right) / 70yrs}$$

(Equation does not show units, see below)

CR	Incremental excess Cancer Risk Level	1 in 100,000 (1×10^{-5})
CSF	Cancer potency Slope Factor (CSF) for linear carcinogens	Pollutant-specific in $(mg/kg\text{-}d)^{-1}$ (sources: MDH and EPA)
ADAF	Age Dependent Adjusted Factor (ADAF) $ADAF_{<2}$ $ADAF_{2to<16}$ $ADAF_{16\,to\,70}$	 10 for birth to less than 2 years 3 for 2 to less than 16 years 1 for 16 to less than 70 years
D	Durations in years for ADAFs	2, 14, and 54
Water Exposure	Drinking Water Intake Rate (DWIR) = MDH $IR_{<2}$ MDH $IR_{2to<16}$ MDH IR_{16+}	 0.137 L/kg-d 0.047 L/kg-d 0.039 L/kg-d
Fish Exposure	Fish Consumption Rates (FCR) $FCR_{<2}$ $FCR_{2to<16}$ FCR_{Adult}	 0.00086 kg/kg-d (0.86 g/kg-d) 0.00055 kg/kg-d (0.55 g/kg-d) 0.00043 kg/kg-d (0.43 g/kg-d)
	Bioaccumulation Factors (BAFs): State BAFs proportioned by Trophic Level (TL) 4 at 76% and TL 3 at 24% & Lipid content 6% for Class 2A and 1.5% TL4 and 2% TL3 for Class 2Bd	Pollutant-specific in L/kg (sources: MPCA and EPA)

Revised IW + FC Linear Cancer Algorithm 2

Beneficial Use Classification

Class B: Fish Consumption and Recreation (Incidental Water exposure), CS_{fr} (µg/L)

$$= \frac{CR\,(1 \times 10^{-5}) \times 1000}{\left\{\left\{CSF \times ADAF_{<2} \times D_{<2} \times [IWR + FCR_{<2} \times (0.24BAF_{TL3} + 0.76BAF_{TL4})]\right\} + \left\{CSF \times ADAF_{2\,to\,<16} \times D_{2\,to\,<16} \times [IWR + FCR_{2\,to\,<16} \times (0.24BAF_{TL3} + 0.76BAF_{TL4})]\right\} + \left\{CSF \times ADAF_{16\,to\,70} \times D_{16\,to\,70} \times [IWR + FCR_{Adult} \times (0.24BAF_{TL3} + 0.76BAF_{TL4})]\right\}\right\} / 70yrs}$$

(Equation does not show units, see below)

CR	Incremental excess Cancer Risk Level	1 in 100,000 (1 x 10^{-5})
CSF	Cancer potency Slope Factor (CSF) for linear carcinogens	Pollutant-specific in $(mg/kg\text{-}d)^{-1}$ (sources: MPCA, MDH and EPA)
ADAF	Age Dependent Adjusted Factor (ADAF) $ADAF_{<2}$ $ADAF_{2to<16}$ $ADAF_{16\,to\,70}$	 10 for birth to less than 2 years 3 for 2 to less than 16 years 1 for 16 to less than 70 years
D	Durations in years for ADAFs	2, 14, and 54
Water Exposure	Incidental Water Intake Rate (IWR)	0.0013 L/kg-d
Fish Exposure	Fish consumption rates (FCR) $FCR_{<2}$ $FCR_{2\,to\,<16}$ FCR_{Adult}	 0.00086 kg/kg-d (0.86 g/kg-d) 0.00055 kg/kg-d (0.55 g/kg-d) 0.00043 kg/kg-d (0.43 g/kg-d)
	Bioaccumulation Factors (BAFs): State BAFs proportioned by Trophic Level (TL) 4 at 76% and TL 3 at 24% & Lipid content 1.5% TL4 and 2% TL3 for Class 2B	Pollutant-specific in L/kg (sources: MPCA and EPA)

New Fish Tissue Linear Cancer Algorithm 2

Beneficial Use Classification

Class 2A, 2Bd, 2B: Fish Consumption, pollutant with a final BAF > 1,000, CS_{ft} (mg/kg)

$$\frac{CR\ (1\ x\ 10^{-5})}{\left[\frac{(CSF x\ ADAF_{<2}\ x\ D_{<2} x\ FCR_{<2}) + (CSF x\ ADAF_{2\ to<16}\ x\ D_{2\ to<16}\ x\ FCR_{2\ to<16}) + (CSF x\ ADAF_{16\ to\ 70} x\ D_{16\ to\ 70}\ x\ FCR_{adult})}{70\ years}\right]}$$

(Equation does not show units, see below)

CSF	Cancer potency Slope Factor (CSF) for linear carcinogens	Pollutant-specific in $(mg/kg\text{-}d)^{-1}$ (sources: MPCA, MDH and EPA)
ADAF	Age Dependent Adjusted Factor (ADAF) $ADAF_{<2}$ $ADAF_{2\,to\,<16}$ $ADAF_{16\,to\,70}$	 10 for birth to less than 2 years 3 for 2 to less than 16 years 1 for 16 to less than 70 years
D	Durations in years for ADAFs	2, 14, and 54
CR	Incremental excess Cancer Risk Level	1 in 100,000 (1 x 10^{-5})
Fish Exposure	Fish consumption rates (FCR) $FCR_{<2}$ $FCR_{2\,to\,<16}$ FCR_{Adult}	 0.00086 kg/kg-d (0.86 g/kg-d) 0.00055 kg/kg-d (0.55 g/kg-d) 0.00043 kg/kg-d (0.43 g/kg-d)

f. Mixtures analysis for human health-based water quality standards

HH-WQS methods currently include specific language to deal with and limit exposure to carcinogens in surface water. The carcinogen additive algorithm ensures exposure to more than one carcinogen in surface waters does not exceed the additional incremental cancer risk level of 1 in 100,000 (Minn. R. 7050.0222, subp. 7, item D, and 7052.0230, subp. 2). Minn. R. ch. 7052 also has general language for noncarcinogens.

To ensure HRLs meet their health protective goal, MDH includes methods based on additivity to ensure carcinogens do not exceed the additional lifetime cancer risk of 1 in 100,000 and noncancer effects remain below appreciable adverse risk. All chemicals found in a groundwater sample are reviewed on the basis of common Health Endpoint(s) for all durations and for cancer when there is a cancer HRL. Health Risk Index–the addition of the hazard quotients of chemicals with similar health endpoints–must not exceed one (*Multiple Chemical HRL* in Minn. R. 4717.7820 and 4717.7870 to 4717.7890).

As introduced, MDH's HRL rule has methods to evaluate multiple chemicals when present in a groundwater sample. The methods stem from two EPA guidance documents on mixtures analysis (USEPA 1986a, USEPA 2000b). There is a hierarchy of options for assessing chemical mixtures based on the extent of information available: ranging from toxicological data on a defined chemical mixture to default application of additivity. Few toxicological evaluations are available for defined mixtures of chemicals; one example was the *Interaction Profile for Persistent Chemicals Found in Fish* (ATSDR 2004). Otherwise, the next two most common approaches for protecting humans from adverse effects stemming from exposure to mixtures of pollutants are to develop toxic equivalency factors (TEFs) for comparing chemicals of similar structure and toxicity profiles (e.g., dioxin-like compound using TEFs based on 2,3,7,8-TCDD, Minn. R. ch. 7052) and to conduct additivity analyses by common MOAs, target organs, or systems, or on cancer (MDH 2008a).

MPCA is proposing to supplement protection to surface water users by including a new approach for noncancer mixtures additivity analysis modeled on the HRL rule. As the revised HH-WQSs already incorporate newer, more robust RfD analysis by MDH, to maintain consistent protection for human health, pollutant mixtures will be addressed by the same approach. The algorithm is based on summing up the ratio of each pollutant concentration measured in the surface water or in fish tissue to their respective CS (or site-specific CC) with the same Health Endpoint. To ensure total exposure does not exceed the threshold for noncancer effects in the target organ, system, or process (development), the sum or Health Risk Index has to be equal to or less than one to meet the HH-WQSs.

$$Noncancer\ Health\ Risk\ Index\ by\ Common\ Health\ Endpoint\ = \frac{C_1}{CS_1\ or\ CC_1} + \frac{C_2}{CS_2\ +\ CC_2} + \cdots + \frac{C_n}{CS_n\ +\ CC_n} \leq 1$$

Where:
$C_1 \ldots C_n$ are the surface water concentrations (as a 30-day average) or fish tissue concentrations for the first through the n^{th} noncancer pollutant with the same *Health Endpoints*.

$CS_1 \ldots CS_n$ (or $CC_1 \ldots CC_n$) is the Chronic Standard (or site-specific Chronic Criterion) for water or fish tissue by use classification (either CS_{dfr}/CC_{dfr} or CS_{fr}/CC_{fr} or CS_{ft}/CC_{ft}) for the first to the n^{th} noncancer pollutant.

For linear carcinogens, the additivity algorithm is as listed in Minn. R. 7050.0222, subp. 7.D. and 7052.0230, subp. 2.

> The additivity equation applies to chemicals that are linear carcinogens and have HH-WQSs calculated with a cancer potency Slope Factor (CSF). For consistency in approaches, this index also can be equal to or less than one. A value greater than one indicates a cancer risk level greater than 1 in 100,000.

$$Cancer\ Health\ Risk\ Index = \frac{C_1}{CS_1 + CC_1} + \frac{C_2}{CS_2 + CC_2} + \cdots + \frac{C_n}{CS_n + CC_n} \leq 1$$

Where:
$C_1 \ldots C_n$ are the concentrations [(as a 30-day average)][41] or fish tissue concentrations for the first through the n^{th} carcinogen.

$CS_1 \ldots CS_n$ or ($CC_1 \ldots CC_n$) is the Chronic Standard (or site-specific Chronic Criterion) for water or fish tissue by use classification (either CS_{dfr}/CC_{dfr} or CS_{fr}/CC_{fr} or CS_{ft}/CC_{ft}) for the first to the n^{th} carcinogen.

C. Guidance options for addressing fish tissue monitoring data and health consultations

a. Use of fish tissue criteria based on new algorithms in conjunction with the narrative fish tissue standards-MDH Fish Consumption Advice

Currently, Minn. R. ch. 7050 has one fish tissue HH-WQS for mercury; otherwise when evaluating fish tissue data for compliance to HH-WQSs MPCA uses the narrative language in Minn. R. 7050.0150, subp. 7 based on MDH Fish Consumption Advice (FCA) (see Sections III. B. and IV.E.). As part of the proposed revisions to Minn. R. chs. 7050 and 7052, MPCA plans to include algorithms to develop fish tissue-based CS_{ft} for statewide application and on a site-specific basis as CC_{ft}. MPCA will update water and add fish tissue HH-WQSs for all listed and new toxic pollutant with a final BAF > 1,000 (definition for BCCs) in future WQS revisions as updated toxicological reviews are completed by MDH or using MDH methods as incorporated into the revised methods for HH-WQS in Minn. R. ch. 7050.

The goal of including fish tissue-based algorithms and HH-WQSs into the rule is ultimately to have these standards replace the reference to FCA. As described in Section IV.E. and recognized by EPA (USEPA 2000a), there are important differences between to the two programs, so while

[41] The revised algorithm will include a reference to the concentration being a 30-day average and use new parameter abbreviations.

the FCA is a tool for determining if HH-WQSs are being met, having actual fish tissue CS_{ft} is the more appropriate threshold for determining if waters are meeting HH-WQSs or not and need to be listed as impaired on the CWA 303(d) list. Just one advantage of not referencing FCA for impaired waters is that while a waterbody may be listed as impaired, for example because of excess mercury in the fish, by following the FCA, fish can still be eaten at amounts that limit the potential of health effects. Impairment listing has a different purpose in identifying waterbodies that need additional measures to reduce pollutant loading; ultimately to reduce the number of limitations on fish consumption.

During the transition time between adopting the new fish tissue algorithms and having CS_{ft} for all pollutants with BAFs > 1,000 that will allow removal of the reference to FCA, MPCA may need to develop CC_{ft} for pollutants detected in fish tissue that lack FCA; any calculated CC_{ft} would be done using current toxicological values reviewed by MDH on a pollutant-specific basis to determine if HH-WQSs are being met in fish tissue. Use of a CC_{ft} would be accomplished through impaired waters list development or other process open for public comment and would require EPA approval if a Site-Specific Modified Standard is needed (Minn. R. 7050.0220, subp.7 or 7052.0270).

b. Role of MDH health consultations

MDH plays a critical role in Minnesota for supporting numerous human health risk-based programs, including public requests for information on toxic chemicals. MDH's Site Assessment and Consultation (SAC) Unit develops site-specific *Health Consultations* as part of a program with the Center for Disease Control's Agency for Toxic Substances and Disease Registry (ATSDR) (more details are available at http://www.health.state.mn.us/divs/eh/hazardous/index.html). MPCA may determine that for particular pollutants and scenarios where contaminants are detected in surface water, a more detailed site-specific review is needed to supplement the information currently available for developing HH-WQSs. MPCA has not refined the criteria to establish when a *Health Consultation* is required. In the past they have been requested to examine ongoing detections of a VOC (trichloroethylene) at a public beach and PAHs in sediments at a Superfund site used for swimming (Minnesota Department of Health (MDH) and Agency for Toxic Substances and Disease Registry 2006, MPCA 2008, MDH 2008b). If a MDH *Health Consultation* determines alternate, more stringent water concentrations are recommended to protect human health, MPCA would proceed to develop a site-specific modified standard requiring EPA approval or site-specific Chronic Criteria (CC) as discussed in Minn. R. chs. 7050 and 7052.

VI. Uncertainty Analysis

A. Introduction

When developing Human Health-based Water Quality Standards (HH-WQSs) or completing any type of human health risk assessment, there is always a level of uncertainty involved. For some aspects of the process, the uncertainty can be quantitatively estimated or has been addressed with standardized inputs (e.g., Uncertainty Factors used to develop Reference Doses). For others, the uncertainty is less well defined and described in qualitative ways (e.g., graded confidence rating from low to high). As described in Risk Characterization, the pollutant properties can also affect the application of HH-WQS as a meaningful tool for protecting human health. Mercury is an example of this limitation, where the main sources and unique environmental fate mean application of the HH-WQS cannot fully protect fish consumers. Public outreach through MDH's Fish Consumption Advice (FCA) program provides the necessary consumption information to protect consumers of fish.

In addition, the implementation of HH-WQSs employs additional levels of protection based on rule language that addresses other aspects of water quality protection that go beyond the numeric WQSs alone (e.g., nondegradation provisions and point source wastewater controls). The implementation of the HH-WQSs though NPDES permitting and assessment of surface water monitoring data to determine if beneficial uses are being met or not combines other considerations with WQSs to provide a multifaceted approach to ensuring protection to all surface water users (MPCA 2012).

MPCA closely reviewed the revised methods with MDH and EPA to ensure they are consistent with the latest science, policy, and public health practices for risk assessment. New science can emerge or has consistent gaps that mean the methods or individual pollutant HH-WQS may not fully meet their goals. However, some conservatism is built into the methods when developing toxicological values and exposure rates to provide a margin of safety.

B. Toxicological evaluation

The toxicological values used in HH-WQSs originate from RfDs and CSFs developed by MDH or EPA's Integrated Risk Information System (IRIS). A key aspect of the revised HH-WQSs is to incorporate better toxicological values based on use of newer animal assays and MOA information. MDH has led the way in applying these newer approaches as part of their HRL rule revisions; MPCA plans to use MDH reviewed toxicological values for future revisions to or for new HH-WQSs or values based on the revised methods planned for adoption into Minn. R. ch. 7050. However, all pollutants with HH-WQSs will not be revised at once, but over time. HH-WQSs based on 1990 algorithms may have greater uncertainty than those based on the revised methods. The accuracy of toxicological data is very pollutant specific; however, the newer methods developed by MDH are more comprehensive and specifically address life stage susceptibility leading to more certainty that revised HH-WQSs meet protection level goals than

past toxicological evaluations. Future HH-WQSs will identify the algorithms used by year adopted (1990, 1998 for GLI adopted into Minn. R. ch. 7052, or revised methods-expected to be final in 2014).

Sources are available that provide more information on how HH-WQSs address uncertainty in toxicological values. MDH described fully the uncertainty associated with their new toxicological evaluations in their 2008 SONAR (Section IV.E.4.). Also of note, MDH will not use a RfD to adopt a HRL if the UF is greater than 3,000 (Minn. R. ch. 4717). The background on uncertainty associated with RfDs and CSFs used as the basis for HH-WQSs adopted prior to these revised methods (2014) are best discussed in historical IRIS Guidance Documents (available at http://www.epa.gov/iris/backgrd.html) or each pollutant's IRIS summary (available at http://cfpub.epa.gov/ncea/iris/index.cfm?fuseaction=iris.showSubstanceList).

C. Exposure evaluation

The HH-WQSs incorporate exposure estimates for drinking water, incidental water ingestion, and fish consumption. Each estimate has uncertainty along with different approaches to address it. MDH again discusses their approach for drinking water intake rates in their 2008 HRL rule SONAR. MPCA developed the revised IWR based on swimming and wading activity patterns under the guidance of defining reasonable maximum exposure (USEPA 1989, MDH and ATSDR2006) and use of newer estimates of incidental ingestion by children (USEPA 2011). The IWR of 0.0013 L/kg-d is higher and more defined to account for toddlers and children that would be most likely to ingest water while swimming than the previous default value of 0.01 L/d or 0.00014 L/kg-day (see Section IV.C.c.). There are limitations on the exposure data for incidental water and general recreational exposure, as described by EPA (USEPA 2011, Section 3.6.1.1.); however, the use of a higher rate and conservative RSC, along with other practices of HH-WQSs development provides assurances that final standards will protect human health.

Developing estimates of exposure from fish consumption involves both the FCRs and BAFs for water HH-WQSs. MPCA developed the adult FCR for the 1990 algorithm based on the best regional surveys completed to date on sport fish intake and patterns for adults (Appendix B1). Since that rate was promulgated only a few surveys have provided data to examine the 30 g/day. As discussed in Section IV.C.e., the available results suggest that 30 g/d was generally in the 80th to 95th percentile of respondents. There are many limitations in these studies (USEPA 1997, USEPA 2000b, USEPA 2008, USEPA 2011), including patterns and demographics of sport fishing population likely have changed in the last 20 years, study participants only occasionally included women and children, or Native American tribal members , and no data on other possible subsistence subpopulations. However, based on the range of survey data and rates used by EPA's and MDH's Fish Consumption Advice programs, the continued use of 30 g/d with a body weight of 70 kg (adult FCR of 0.43 g/kg-d)as the basis for a chronic FCR remains representative of a higher percentile exposure estimate for most freshwater, caught fish consumers (USEPA 1997, USEPA 2000b, USEPA 2011). Information is also provided for the development of alternate FCRs in scenarios where the default rates can be improved upon with site- or pollutant-specific data.

Populations of subsistence fishers may consume more fish; in the case of available and reliable data MPCA can modify fish consumption rates when requested on a site-specific basis as described in Minn. R. chs. 7050 and 7052. Tribes can also develop their own WQSs and two Minnesota Chippewa Tribes in the Lake Superior Basin have done so using higher fish consumption rates. In addition, as described in *Risk Characterization*, HH-WQSs include additional measures to help ensure they meet the health protective goals, which is not based on intakes rates alone, but all the parameters that go into HH-WQS algorithms.

One such additional measure is MPCA's review of survey data and development of a new children's FCR (Maschwitz and Preimesberger 2009). As described in the article and Section IV.C.e., there were definite data limitations that precluded the development of a children's fish intake rate in a manner consistent with the adult rate. There is, however, sufficient data to estimate the ages at which children's fish intake may be greater than an adult's on a per body weight basis (ages 1 through 5) and propose the use of a children's FCR that is twice the adult's rate. The FCR of 0.86 g/kg-d is based on a "child to adult ratio" of two observed in a number of survey statistics and also reflects an advancement of MPCA policy to add more protection for vulnerable developmental life stages in the linear cancer algorithms.

Bioaccumulation Factors (BAFs) account for the accumulation of a pollutant in fish tissue and are the other important parameters required to estimate exposure from fish consumption when setting water-based HH-WQSs. The revised BAF methods for statewide application reflect newer science already implemented for Lake Superior HH-WQSs. BAFs are highly dependent on a number of factors including the pollutant characteristics, type of BAF studies, fish species consumed, and additional ambient water quality characteristics. EPA's BAF methods can reasonably address many of these factors, but the variability and uncertainty in developing BAFs is also represented in their definition: represent long-term, steady-state, average bioaccumulation. One of EPA's BAF guidance documents summarizes the uncertainty in these values (Table 3-1 in USEPA 2003b). Depending on the pollutant's physicochemical characteristics, BAFs may be over or under estimated. For pollutant's with BAFs greater than 1,000 (e.g., GLI BCCs), this issue is addressed through the application of fish tissue-based HH-WQSs, which are a more accurate application of standards for these pollutants than water standards. Conversely, expanding the data reviewed for pollutants with less field data will improve the accuracy of BAFs, toxicological evaluations, and stated relevance of the fish pathway (see Section IV.E.)

As part of the noncancer or nonlinear carcinogen algorithms, a RSC is applied to account for other noningestion water and nonwater routes of exposure; the goal of using the RSC is to ensure a person's average daily exposure to a pollutant remains below the RfD (Section IV.C.h.). EPA's Exposure Decision Tree has included a few more refinements for developing a RSC; but like the previous approaches, limitations on actual exposure data for most pollutants means use of default values. The uses of the 0.2, or for some classes of pollutants 0.5, are generally considered reasonable for limiting total exposure below adverse levels (USEPA 2000b).

D. Risk characterization

Many factors come together to help ensure HH-WQSs meet their *Protection Level Goals*. The proposed revised parameters and algorithms offer many improvements from the past, with more accurate exposure estimates and toxicological values. In addition, the application of more protection for developmental life stages will result in more meaningful "lifetime" protection from surface water pollutants (additional discussion is found in USEPA 2000b). Where there are uncertainties in risk assessment approaches to address site-specific scenarios that may have different risk than used to calculated HH-WQSs, there are new guidance options for better using consultations with MDH to modify standards.

Steps have been taken by MPCA to strengthen narrative standards and pollutant-specific HH-WQSs. The revised methods add approaches for addressing degradates of pollutants and mixtures of pollutants. There is uncertainty in both proposed approaches (e.g., degradates more toxic than parent pollutant or synergy of mixtures) (MDH 2008); however, inclusion of more defined approaches for these pollutants improves the current methods. MPCA will continue to follow advancements in risk methods and pollutant toxicological profiles for future revisions; however, in the meantime, the revised methods allow for use of more accurate chemical-specific inputs over defaults when adequate data are available.

Overall, HH-WQSs serve to protect beneficial uses of surface waters for human users and to generally limit that population's exposure to toxic pollutants. In the broader context of environmental pollutant management, HH-WQSs provide a meaningful approach in one media; however, there are limitations to single media standards. For example, some chemicals because of their wide range of physicochemical properties can be detected or discharged into surface waters, but are more often present in air and require standards or management options in that media to more accurately address. Therefore, while many VOCs were identified in the Clean Water Act (CWA) as *Priority Pollutants*, methods are not refined to standardize inclusion of the inhalation pathway, so the standard WQS algorithms are built on incidental water and fish consumption with use of RSCs or stringent CR levels to address uncertainty. MPCA will also seek to complete site-specific MDH *Health Consultations* for scenarios where more accurate guidance values would be needed for greater protection. In contrast to VOCs, BCCs, such as dieldrin, where the fish pathway is a principal route of exposure, the accuracy of the HH-WQSs will be increased and they will provide a more precise level of protection based on current science and revised algorithms. These differences in properties also mean the context for pollution control has more or less relevance under the CWA and WQS regulations. Air sources are known to be much more significant sources to surface waters and subsequently fish than direct dischargers for many BCCs, such as mercury and PCBs. While the HH-WQSs may not provide a specific mechanism for controlling an air pollutant, they do provide goals for pollution management in air programs (e.g., mercury reductions in products and air emissions).

VII. Details of Class 1 and Class 2 Chronic Standard Applications

A. Class 1 domestic consumption standards

Class 1 Domestic Consumption (DC) standards are the Safe Drinking Water Act (SDWA) primary (MCLs) and secondary standards incorporated by reference, except for some pollutant classes as listed in Minn. R. 7050.0220, subp. 2.A, and 7050.0221, subp. 1.B. The MCLs, as applied by the SDWA, except those for nitrate and nitrite, are based on chronic protection to drinking water users and are applied as annual-average concentrations to finished drinking water. In contrast, nitrate and nitrite are acutely toxic to infants, who can develop methemoglobinemia (blue baby syndrome). The duration applied under the SDWA for nitrate and nitrite is based on not exceeding 10 and 1 mg/L, respectively, after averaging two samples in a 48-hour window following a single high sample concentration.

Minn. R. ch. 7050 applies the DC standards in untreated surface waters and groundwater. These standards have been incorporated by reference without specific language on their application in groundwater or surface waters. Application of DC standards in ambient surface waters needs to consider their role in public drinking water protection, with Class 2 HH-WQSs, and for point source effluent limits.

MPCA initiated the first impaired surface waters listings for nitrate using the Class 1 DC standard in the 2010 impaired waters (303d list). The MCL for nitrate of 10 mg/L was applied as a one-day average concentration. The assessment protocol considers frequency in determining if a waterbody is not meeting WQS and is listed as impaired. When monitoring data show two, one-day average concentrations above the DC standard in a three-year window that waterbody was considered in violation of the Class 1standard and listed as impaired.

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

Period of Record	Minimum No. of Data Points [Depends on Dataset]	Use Support or Listing Category Based on Exceedances of Drinking Consumption Standard (10 mg/L)	
DC Acute Standard* Exceedance Threshold →		No more than 1 in 3 yrs.	2 or more in 3 yrs.
Most recent 10 yrs.	5, within a 3-yr period	Not Listed	Listed

*24 –hour central value

For all other Class 1 DC standards based on the MCLs and Secondary Drinking Water Standards, their application has been program specific. MPCA is considering clarifying application in a future rulemaking effort.

B. Class 2 aquatic life and human health-based WQSs

a. Water-based Chronic Standards

As introduced in the beginning of this document, long-term or chronic protection of surface waters includes beneficial uses for humans and aquatic communities. WQSs based on aquatic life toxicity cover acute and chronic durations. The acute WQSs: Final Acute Value (FAV) and Maximum Standard (MS) are always listed in rule; the MS protects aquatic life from short-term excursions of the CS and is applied as a one-day average concentration. The averaging period for a CS_{tox} based on direct aquatic life toxicity is four days as recommended by EPA (USEPA 1994). The four-day averaging period is based on the short life cycle of some organisms, and the fact that chronic toxicity in fish is often defined by the sensitivity of a life stage that lasts only a few days.

The averaging period for standards based on human health or fish-eating wildlife is 30 days (Minn. R. 7050.0222. subp. 7). The longer averaging period is used because these standards assume lifetime exposure to the chemical, approximates the time it takes many bioaccumulative pollutants to reach steady-state in aquatic organisms, and is practically implemented for effluent limits.

In addition, as part of the revised methods and use of less-than-chronic durations, 30 days matches the definition of *short-term* used for the protection of drinking water as part of the *Drinking Water Supplemental Algorithm*. Therefore, continued use of the 30-day application duration provides a consistent and more protective averaging time when the final CS is based on short-term or subchronic adverse effects. Based on the chemicals reviewed to-date by MDH in the HRL program, no acute (one day) duration values were found to result in more stringent concentrations than those calculated for short-term to chronic, providing additional rationale to maintain the 30-day averaging time for all CS based on human health. If there are chemicals reviewed in the future and adopted as HH-WQSs (CSs) or site-specific CC that demonstrate a one-day or alternate averaging time less than 30-days is needed to be sufficiently protective, MPCA would apply an alternate averaging time.

Application of the CS has typically been approached as the "lowest single CS calculated for:

1. The CS based on toxicity to aquatic organisms, animals or plants, CS_{tox};
2. The CS based on human health, including replacement with an EPA organoleptic standards (concentration that will prevent unacceptable taste or odor in water, fish, or shellfish) if more stringent, $CS_{dfr}/CS_{fr}/CS_{dev,}$ Or,
3. In Minn. R. ch. 7052, CS_w, wildlife-based CS to protect mammals and birds that eat aquatic organisms.

However, MPCA recognizes that between aquatic toxicity-based CSs (for either animal or plants) and human health-based (or fish-eating wildlife in Minn. R. ch. 7052), even if one value is more stringent, the difference in averaging times may mean that CSs are actually complementary and should all be considered "applicable." MPCA will only develop one aquatic toxicity-based and one human health-based CS for listing as applicable.

Besides magnitude (numeric standard) and duration, application of Class 2 CSs applied in water also has a frequency component. MPCA applies a three-year window when applying the CS as shown in Table 8. The protection level built into CS is considered stringent enough to allow an exceedance once in a three-year window and still be protective of beneficial uses. In addition, ambient CSs applied in surface water to protect drinking water use are one-step removed from actual exposure from treated drinking water regulated through the Safe Drinking Water Act program at EPA and MDH. The SDWA standards for toxic standards are applied as longer, annual averages, except for nitrate and nitrite. However, two such exceedances of a human health-based CS in a three-year window are more likely to result in harmful effects and is evidence of impairment of the HH-WQSs. MPCA assessment of monitoring data can also include other factors when determining impairment, such as the magnitude of exceedance, mixtures of similar acting toxic pollutants, number of data points, and applicability of historical data for each assessment cycle. Final decisions on assessment of a human health-based CS based on professional judgment would be described in the impaired waters assessment guidance that accompanies each CWA 303(d) list of impaired waters. MPCA takes public comment on this guidance and list every two years.

Table 8: Summary of data requirements and exceedance thresholds for assessment of pollutants with human health-based and wildlife-based standards

Period of Record	Minimum No. of Data Points [Depends on Dataset]	Use Support or Listing Category Based on Exceedances of Chronic Standard	
Chronic Standard* Exceedance Threshold →		No more than 1 in 3 yrs.	2 or more in 3 yrs.
Most recent 10 yrs.	5, within a 3-yr period	Not Listed	Listed

*30 –day central value

b. Fish Tissue-based Chronic Standards

Fish Tissue-based Chronic Standards (CS_{ft}) are not applied with durations or frequency considerations. As described in MPCA's impaired waters guidance for mercury (MPCA 2012), MPCA has developed waterbody assessments for mercury in fish tissue data based on consistency with the 2007 Statewide Mercury Total Maximum Daily Load (TMDL) study[42]. For other pollutants–currently applied to PCBs and PFOS–the MDH Fish Consumption Advice site-specific listings (see http://www.health.state.mn.us/divs/eh/fish/eating/sitespecific.html) are used as the foundation for impairment. If any fish species has FCA more restrictive than one meal per week (based on the narrative standard in Minn. R. ch. 7050.0150), that waterbody is listed as impaired.

As CS_{ft}s are adopted those numeric standards will be used to assess edible fish tissue monitoring data, unless the narrative standard is more stringent (Minn. R. 7050.0150, subp. 2). MPCA will eventually remove the narrative linking impairment to MDH FCA after adopting CS_{ft} for all currently promulgated HH-WQS that also meet the BCC definition. When developing a site-specific CC_{ft} MPCA will consult with MDH on appropriate toxicological values for consistency in these values; however, based on applying the methods and algorithms in Minn.

[42] Details of the mercury TMDL and subsequent reduction efforts is found at http://www.pca.state.mn.us/wfhy9ef.

R. 7050.0218 (and 7050.0219 planned for adoption), the final criteria can result in different numeric thresholds than MDH FCA or may be developed for a pollutant lacking MDH FCA. MPCA will consider current and historical analytical methods, past and current impairment status, and documented trends of that pollutant in fish tissue to determine the relevancy and use of historical monitoring data in assessments (i.e., mercury data are applicable going back 10 years). MPCA plans to apply the CS_{ft} to any fish species with acceptable pollutant monitoring data, with preference for collecting fish species and sizes regularly consumed by human.

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APPENDIX A1— Definitions for Human Health Based Chronic Standards (Inclusion in Minn. R. chs. 7050 & 7052 Determined with Rule Language Process)

Revised, 2012

Pertinent definitions and abbreviations are listed below. The text in bold represents new rule language being proposed for Minn. R. ch. 7050 and 7052. The source of definitions when taken from other documents is indicated.

Adverse effect – A biochemical change, functional impairment, or pathologic lesion that affects the performance of the whole organism or reduces an organisms' ability to respond to an additional environmental challenge (MDH 2008a).

Age-dependent adjustment factor (ADAF) – Numerical multipliers used to modify the cancer slope factor that account for the increased susceptibility to cancer from early-life exposures to linear carcinogens in the absence of chemical-specific data (Minn. R. 4717.7820, subp. 3). For default use there are three ADAFs:

$ADAF_{<2}$ = 10, for birth up to two years of age; $ADAF_{2\ to\ <16}$ = 3, for two up to 16 years of age; and $ADAF_{16+}$ = 1, for 16 years of age and older.

$AF_{Lifetime}$ or lifetime adjustment factor – Numerical multiplier used to modify the adult-based cancer slope factor for lifetime (70 years standard duration in risk characterization) exposure based on chemical-specific data (Minn. R. 4717.7820, subp. 2).

Available **and reliable** scientific data – The information derived from scientific literature including: published literature in peer reviewed scientific journals, USEPA ambient water quality criteria documents, and other reports or documents published by the USEPA or other governmental agencies.

Benchmark dose (BMD) – Dose or concentration that produces a predetermined change in the response rate of an adverse or biologically meaningful effect. The BMD approach uses mathematical models to statistically determine a dose associated with a predefined effect level (e.g., 10 percent). BMDL: A statistical lower confidence limit on the BMD (MDH 2008a).

Bioaccumulation factor (BAF) - The concentration of a pollutant in one or more tissues of an aquatic organism, exposed from any source of the pollutant but primarily from the water, diet, and bottom sediments, divided by the average concentration in the solution in which the organism had been living, under steady state conditions.

Bioaccumulative chemical of concern (BCC) – Any chemical that has the potential to cause adverse effects which, upon entering the surface waters of the state, by itself or as its toxic transformation product, accumulates in aquatic organisms by a human health bioaccumulation

factor (BAF) greater than 1,000, after considering metabolism and other physiochemical properties that might enhance or inhibit bioaccumulation, in accordance with the methodology in part 7052.0110, subpart 3.

Bioconcentration factor (BCF) - The concentration of a pollutant in one or more tissues of an aquatic organism, exposed only to the water as the source of the pollutant, divided by the average concentration in the solution in which the organism had been living, under steady state conditions.

Biomagnification - An increase in the tissue concentration of a pollutant in aquatic organisms at successive levels trophic levels through a series of predatory-prey associations, primarily occurring through dietary accumulation. The expression used to quantify this increase is the Biomagnification Factor (BMF). For a given water body the BMF is calculated as: 1) The ratio of the tissue concentration of a pollutant in a predator at a particular trophic level to the tissue concentration in the its prey at the next lower trophic level; or 2) The ratio estimated from a comparable laboratory model (USEPA 2000b).

Biota-sediment accumulation factor (BSAF) – The ratio in (kg of organic carbon/ kg of lipid) of a pollutant's lipid-normalized concentration in tissue of an aquatic organism to its organic carbon-normalized concentration in surface sediment, in situations where: the ratio does not change substantially over time, both the organism and its food are exposed, and the surface sediment is representative of average surface sediment in the vicinity of the organism.

Cancer potency **slope** factor (C**SF**) - A factor indicative of a chemical's human cancer causing potential. **An upper-bound estimate of cancer risk per increment of dose that can be used to estimate cancer risk probabilities for different exposure levels (Minn. R. 4717.7820, subp. 23).** The CSF is expressed in **units of cancer incidence per milligram of pollutant per kilogram of body weight per day** (**mg/kg-day**)$^{-1}$(previous acronym q1*).

Cancer Risk Level (CR) – Probability that daily exposure to a carcinogen over a lifetime may induce cancer (Minn. R. 4717.7820, subp. 4); use in this rule refers to an incremental or additional excess cancer risk equal to 1 x 10^{-5} (1 in 100,000) and is applied with the cancer potency slope factor for single chemicals and for mixtures as described in Minn. R. 7050.0222, subp. 7.D.

Carcinogen – a chemical:

a) **Classified as a human carcinogen or probable human carcinogen according to the "EPA Classification System for Categorizing Weight of Evidence for Carcinogenicity from Human and Animal Studies," the Risk Assessment Guidelines of 1986, United States Environmental Protection Agency, Office of Health and Environmental Assessment (August 1987);**
b) **Classified as "carcinogenic to humans" or "likely to be carcinogenic to humans" according to the Final Guidelines for Carcinogenic Risk Assessment, United States Environmental Protection Agency, Office of Research and Development (March 2005). The final guidelines are available at http://www.epa.gov/cancerguidelines/; or**
c) **Classified as a chemical known to be a human carcinogen or reasonably anticipated to be a human carcinogen in the Report on Carcinogens, United States Department of**

Health and Human Services, Public Health Service, National Toxicology Program. The report is available at http://ntp.niehs.nih.gov/go/roc. (Minn. R. 4717.7820, subp. 5)

Carcinogen, linear (C) – Chemical for which, either by a known mode of action or a conservative assumption, the associated cancer risk varies in direct proportion to the extent of exposure, and for which there is no risk-free level of exposure (MDH 2008a). The toxicological value for a linear carcinogen is the cancer potency slope factor (CSF). Seventy years is the standard lifetime duration utilized by the USEPA in the characterization of lifetime cancer risk.

Carcinogen, nonlinear (NLC) - Chemical, for which, particularly at low doses, the associated cancer risk does not rise in direct proportion to the extent of exposure, and for which a threshold level of exposure exists below which there is no cancer risk (Minn. R. 4717.7820, subp. 20). For nonlinear carcinogens, the reference dose (RfD) is the toxicological value used as the threshold for cancer risk.

Chronic Toxicity - A stimulus that lingers or continues for a long period of time, often one-tenth the life span or more. A chronic effect can be mortality, reduced growth, reproduction impairment, harmful changes in behavior, and other non-lethal effects.

Chronic criterion (CC) and chronic standard (CS) - The highest water concentration **or fish tissue concentration** of a toxicant or effluent to which organisms: **aquatic life, humans, or wildlife** can be exposed **indefinitely** without causing chronic toxicity. **CC represents a site-specific chronic criterion developed based on this part (7050.0218) and 7050.0219 or 7052.0110. CS represents a chronic standard listed in parts 7050.0220 and 7050.0222 or in Minn. R. 7052.0100. CC and CS are further distinguished by the organisms they are developed to protect and media they apply:**

1) **CC_{tox} or CS_{tox} represent values applied in surface water developed to protect aquatic life from chronic toxicity**.
2) CC_{dfr} **or** CS_{dfr} **represent values applied in surface water** based on protecting humans from exposure to the pollutant from drinking water, eating fish, **and aquatic recreation**.
3) **CC_{fr} or CS_{fr}-represent values applied in surface water** based on protecting humans from exposure to the pollutant from eating fish **and aquatic recreation**.
4) **CC_{ft} or CS_{ft} represent the values applied in fish tissue to protect humans from exposure to the pollutant from eating fish**.
5) CC_w represent a chronic criterion **applied in surface water** based on protecting wildlife from exposure to the pollutant from eating aquatic organisms. Chronic standards are listed in parts 7050.0220 and 7050.0222 **and in Minn. R. 7052.0100.**

Note: Previous definitions listed "sport-caught" fish; that modifier to "eating fish" was removed, because it wasn't accurate in conveying the basis for new children's fish consumption rates and applicability to a broader range of fish consumers that HH-WQs are designed to protect.

Criterion- A number or numbers established for a pollutant derived under this part (as described in rule) **or in 7052.0110**, or issued by the USEPA, to protect aquatic life, humans, or wildlife.

Developmental health endpoint or developmental toxicity – An adverse effect on the developing organism that may result from parental exposure prior to conception, maternal exposure during prenatal development, or direct exposure postnatally until the time of sexual maturation. Developmental toxicity may be detected at any point in the lifespan of the organism. The major manifestations of developmental toxicity include: A. Death of developing organism, B. Structural abnormality, C. Altered growth, and/or D. Functional deficiency (Minn. R. 4717.7820, subp.8).

Note: The addition of "or" to this definition was recommended by MDH, because developmental toxicity is described by each of the items alone or in combination.

Dose-response assessment – Determination of the relationship between the magnitude of administered, applied, or internal dose and a specific biological response. Response can be expressed as measured or observed incidence, percent response in groups of subjects (or populations), or the probability of occurrence of a response in a population (MDH 2008a). Dose is expressed in units of milligrams of the chemical per kilogram of body weight per day (mg/kg-d).

Dosimetric adjustment factor (DAF) – A multiplicative factor used to adjust observed experimental or epidemiological data to human equivalent concentration for assumed ambient scenario (MDH 2008a).

Duration - The time over which the instream concentration of a pollutant is averaged for comparison with the water quality standards.

Durations for human health-based algorithms (D) – Length of the exposure period under consideration for noncancer and linear cancer algorithms.

1) **The four default durations used in developing RfDs and corresponding Intake Rates are: a) acute: a period of 24 hours or less; b) short-term: a period of more than 24 hours, up to 30 days; c) subchronic: a period of more than 30 days, up to eight years based on application of less than ten percent the standard life expectancy of 70 years for human; or d) chronic: a period of more than eight years.**
2) **The default durations for use in the linear cancer algorithms with ADAFs are: two-years (2) for the birth up to two-years age group; 14-years (14) for the two up to 16-year age group; and 54-years (54) for the 16 up to 70-year age group.**

For any algorithm, use of chemical-specific data to define durations will be preferred when acceptable data are available (Minn. R. ch. 4717.7820, subp. 9.A).

Endocrine (E) – A change in circulating hormone levels or interactions with hormone receptors, regardless of the organ or organ system affected. Health Endpoints with or without the (E) designation are deemed equivalent, for example, thyroid (E) = thyroid, and must be included in the same Health Risk Index Equation (Minn. R. 4717.7820, subp. 10).

Food Chain Multiplier– The ratio of a bioaccumulation factor (BAF) by trophic level to an appropriate bioconcentration factor (BCF) (USEPA 2000b). FCM refers to values developed using USEPA models or from available and reliable field studies.

Frequency - The number of times a standard can be exceeded in a specified period of time without causing acute or chronic toxic effects on the aquatic community, human health, or fish-eating wildlife.

Health Risk Index – Sum of the quotients calculated by identifying all chemicals that share a common Health Endpoint and dividing the water or fish tissue concentration for each chemical (measured or statistically derived) by its applicable CC or CS (Minn. R. 4717.7820, subp. 11). To meet the objectives in part 7050.0217, the health risk index should not exceed a value of one.

Health Risk Index endpoint or health endpoint – General description of toxic effects used to group chemicals for the purpose of calculating a Health Risk Index (Minn. R. 4717.7820, subp. 12).

Human equivalent dose (HED) – Human dose (for other than the inhalation route of exposure) of a chemical that is believed to induce the same magnitude of toxic effect as the experimental animal species dose. This adjustment may incorporate toxicokinetics information specific to the agent, if available, or use a default procedure, such as assuming that daily oral doses experienced for a lifetime are proportional to body weight raised to the 0.75 power ($BW^{3/4}$).

Intake rate (IR) – Rate of ingestion, inhalation, or dermal contact, depending on the route of exposure, expressed as the amount of media taken in, on a per body weight and daily basis, for a specified duration (MDH 2008a). In this rule, the route of exposure is ingestion; media is surface water or fish; and the durations are acute to chronic.

Lowest observable adverse effect level (LOAEL) - **Lowest exposure level at which a statistically or biologically significant increase in the frequency or severity of adverse effects was observed between the exposed population and its appropriate control group (MDH 2008a).**

Magnitude - The acceptable amount of a toxic pollutant in water expressed as a concentration.

Mechanism of action- Complete sequence of biological events (i.e., including toxicokinetics and toxicodynamics events) from exposure to the chemical to the ultimate cellular and molecular consequences of chemical exposure that are required in order to produce the toxic effect. However, events that are coincident but not required to produce the toxic outcome are not included (MDH 2008a).

Mode of action – Sequence of key event(s) following chemical exposure upon which the toxic outcome depends (MDH 2008a); contrast to Mechanism of Action.

No observable adverse effect level (NOAEL) - The highest exposure level at

which there is no statistically or biologically significant increase in the frequency or severity of adverse effects between the exposed population and its appropriate control group (MDH 2008a).

Octanol to water partition coefficient (K_{ow}) – Ratio of the concentration of a substance in the octanol phase to its concentration in the aqueous phase of a two-phase octanol to water system after equilibrium of the substance between the two phases has been achieved. The $\log_{10} K_{ow}$ has been shown to be proportional to the bioconcentration potential of lipophilic organic chemicals.

Point of departure (POD) – Dose-response point that marks the beginning of a low-dose extrapolation. This point can be the lower bound on a dose for an estimated incidence or a change in response level from a dose-response model (BMD) or a NOAEL or LOAEL for an observed incidence, or change in level of response (MDH 2008a).

Priority pollutants - A list of toxic pollutants established under Section 307(a) (1) of the Clean Water Act. As of July 1993, the list included 126 toxic substances.

Reference dose (RfD) - Estimate of a **dose for a given duration** to the human population, including **susceptible** subgroups **such as infants, that** is likely to be without an appreciable risk of adverse effects **during** a lifetime. **It is derived from a suitable dose level at which there is few or no statistically or biologically significant increases in the frequency or severity of an adverse effect between the dosed population and its associated control group.**

The RfD includes one or more divisors, applied to the suitable dose level, accounting for: (i) uncertainty in extrapolating from mammalian laboratory animal data to humans; (ii) variation in toxicological sensitivity among individuals in the human population; (iii) uncertainty in extrapolating from effects observed in a short-term study to effects of long-term exposure; (iv) uncertainty in using a study in which health effects were found at all doses tested; and (v) uncertainty associated with deficiencies in the available data (Minn. R. 4717.7820, subp. 21)[43]. **The product of the divisors is not to exceed 3,000 in an RfD used for a chronic standard.** The RfD is expressed in units of daily dose **as milligrams of chemical per kilogram of body weight-day or** mg/kg-d.

Relative source contribution factor (RSC) – **Percentage or apportioned amount (subtraction method) of the Reference Dose (RfD) for a pollutant allocated to surface water exposures from drinking or incidental water ingestion and fish consumption**. The default values, in the absence of sufficient **pollutant**-specific data, are 0.2 and **0.5**.

Standard - A concentration of a pollutant, established at a level to protect a specified beneficial use that has been adopted into administrative rules.

Time-weighted average (TWA) – When quantifying a measurement that varies over time, such as water intake, a time-weighted average takes measured intakes, which may occur at unevenly spaced intervals, and multiplies each measurement by the length of its interval.

[43] MDH does not derive a promulgated HRL if the uncertainty factors used to derive the RfD exceed 3000 (Minn. R. ch. 4717).

These individual weighted values are then summed and divided by the total length of all of the individual intervals. The result is an average of all of the measurements, with each measurement carrying more or less weight in proportion to its size (MDH 2008a).

Toxic effect – Observable or measurable adverse biological event manifested in an organ, tissue, or system. The designation of endpoints does not exclude other possible observable or measurable biological events. For the purpose of grouping chemicals and creating a health risk index when multiple chemicals are present, toxic effects may be ascribed to more general health risk index endpoints or health endpoints (Minn. R. 4717.7820, subp. 24).

Toxic pollutant - Pollutants or combinations of pollutants, including disease-causing agents, which after discharge and upon exposure, ingestion, inhalation or assimilation into any organism, either directly from the environment or indirectly by ingestion through food chains, will, on the basis of information available to the MPCA, cause death, disease, behavioral abnormalities, cancer, genetic mutations, physiological malfunctions, including malfunctions in reproduction, or physical deformation, in such organisms or their offspring (Minnesota Statutes Chapter 115.01).

Toxicokinetics – Determination and quantification of the time course of adsorption, distribution, metabolism, and excretion of chemicals (sometimes referred to as pharmacokinetics) (USEPA 2010b).

APPENDIX A2-Definitions Comparison: Minn. R. chs. 7050 & 7052; Minn. R. ch. 4717; and Federal Regulations and Guidance (see Appendix A1 for Definitions Proposed for Minn. R. chs. 7050 and 7052)

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
Acceptable daily exposure (ADE)			Estimate of the maximum daily dose of a substance which is not expected to result in adverse noncancer effects to the general human population, including sensitive subgroups		
Acute toxicity (duration)	(Aquatic Life) a stimulus severe enough to rapidly induce a response. In toxicity tests, a response is normally observed in 96 hours or less. Acute effects are often measured in terms of mortality or other debilitating effects, represented as LC50s or EC50s, and expressed as concentrations of mass per unit volume, percent effluent, or toxic units.	(Aquatic Life) stimulus severe enough to rapidly induce a response. In toxicity tests, a response is normally observed in 96 hours or less. Acute effects are often measured in terms of mortality or other debilitating effects, represented as LC50s or EC50s, and expressed as concentrations of mass per unit volume, percent effluent, or toxic units.	(Aquatic Life) concurrent and delayed adverse effect(s) that results from an acute exposure and occurs within any short observation period which begins when the exposure begins, may extend beyond the exposure period, and usually does not constitute a substantial portion of the life span of the organism	A period of 24 hours or less	Studies of acute exposure (one dose or multiple dose exposure occurring within a short time (e.g. less than 24 hours)) are widely available for many chemicals. Acute toxicity [often expressed in terms of the lethal dose (or concentration) to 50 percent of the population (LD50 or LC50)] is usually the initial step in experimental assessment and evaluation of a chemical's toxic characteristics.
Additional lifetime cancer risk	See Incremental Cancer Risk		See Incremental Cancer Risk	The probability that daily exposure to a carcinogen over a lifetime may induce cancer. The Department of Health uses an additional	Target Risk: Value typically in the range of 10-4 to 10-6 **IRIS: Excess Lifetime Risk:** The additional or extra risk of developing cancer due to

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
				cancer risk of 1x10-5 (1 in 100,000) to derive cancer HRLs	exposure to a toxic substance incurred over the lifetime of an individual.
Adverse Effect				A biochemical change, functional impairment, or pathologic lesion that affects the performance of the whole organism or reduces an organism's ability to respond to an additional environmental challenge.	The determination of whether an effect is adverse requires professional judgment. Generally, adverse health effects are considered to be those deleterious effects which are or may become debilitating, harmful, or toxic to the normal functions of an organism, including reproductive and developmental effects. Adverse effects do not include such effects as tissue discoloration without histological or biochemical effects, or the induction of the enzymes involved in the metabolism of the substance.
Age-Dependent Adjustment Factor (ADAF)				Default modifiers to the cancer slope factor that account for the increased susceptibility to cancer from early-life exposures to linear carcinogens in the absence of chemical-specific data. For the default derivation of cancer HRLs, the following ADAFs and corresponding age groups are utilized: $ADAF_{<2}$=10, for birth until	

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
				two years of age; $ADAF_{2\ to\ <16}$=3, for two up to 16 years of age; and $ADAF_{16+}$ =1, for 16 years of age and older.	
Bioaccumulation factor (BAF)	Concentration of a pollutant in one or more tissues of an aquatic organism, exposed from any source of the pollutant but primarily from the water, diet, and bottom sediments, divided by the average concentration in the solution in which the organism had been living, under steady state conditions		Ratio (in L/kg) of a substance's concentration in tissue of an aquatic organism to its concentration in the ambient water, in situations where both the organism and its food are exposed and the ratio does not change substantially over time		The ratio (in liters per kilogram of tissue) of the concentration of a chemical in the tissue of an aquatic organism to its concentration in water, in situations where both the organism and its food are exposed and *the ratio does not change substantially over time*.
Bioconcentration factor (BCF)	Concentration of a pollutant in one or more tissues of an aquatic organism, exposed only to the water as the source of the pollutant, divided by the average concentration in the solution in which the organism had been living, under steady state conditions		Ratio (in L/kg) of a substance's concentration in tissue of an aquatic organism to its concentration in the ambient water, in situations where the organism is exposed through the water only and the ratio does not change substantially over time		The ratio (in liters per kilogram of tissue) of the concentration of a chemical in the tissue of an aquatic organism to its concentration in water, in situations where the organism is exposed through the water only and *the ratio does not change substantially over time*.
Biota-sediment accumulation factor (BSAF)		Ratio (in kg of organic carbon/kg of lipid) of a substance's lipid-normalized concentration in tissue of an aquatic organism to its organic carbon-normalized concentration in surface	Ratio (in kg of organic carbon/kg of lipid) of a substance's lipid-normalized concentration in tissue of an aquatic organism to its organic carbon-normalized concentration in surface		For nonionic organic chemicals (and certain ionic organic chemicals to which similar lipid and organic carbon partitioning behavior applies), the BSAF is the ratio (in kilograms of sediment organic carbon per kilogram

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
		sediment, in situations where the ratio does not change substantially over time, both the organism and its food are exposed, and the surface sediment is representative of average surface sediment in the vicinity of the organism.	sediment, in situations where the ratio does not change substantially over time, both the organism and its food are exposed, and the surface sediment is representative of average surface sediment in the vicinity of the organism.		of lipid) of the lipid-normalized concentration of a chemical in tissue of an aquatic organism to its organic carbon-normalized concentration in surface sediment, in situations where the ratio does not change substantially over time, both the organism and its food are exposed, and the surface sediment is representative of average surface sediment in the vicinity of the organism.
Cancer potency factor (ql*)	A factor indicative of a chemical's human cancer causing potential; the upper 95 percent confidence limit (one-sided) of the slope from a linear nonthreshold dose-response model used by the USEPA to provide an upper bound estimate of incremental cancer risk; assumes a lifetime exposure; expressed in days times kilogram body weight per milligram toxicant (d x kg/mg).		See Slope Factor	See Slope Factor	The cancer potency factor expresses incremental, lifetime risk as a function of the rate of intake of the contaminant.
Carcinogen				"Nonlinear carcinogen": means a chemical agent, for which, particularly at low doses, the associated cancer risk does not rise in direct proportion to the	

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
				extent of exposure, and for which a threshold level of exposure exists below which there is no cancer risk.	
Chronic toxicity (duration)	Stimulus that lingers or continues for a long period of time: ≥ 1/10 life span; effect can be mortality, reduced growth, reproduction impairment, harmful changes in behavior, and other nonlethal effects	Stimulus that lingers or continues for a long period of time; ≥ 1/10 life span; effect can be mortality, reduced growth, reproduction impairment, harmful changes in behavior, and other nonlethal effects	(132.2)concurrent and delayed adverse effect(s) that occurs only as a result of a chronic exposure (appD) adverse effect that is measured by assessing an acceptable endpoint, and results from continual exposure over several generations, or at least over a significant part of the test species' projected life span or life stage.	A period of more than approximately 10% of the life span in humans (more than approximately 90 days to 2 years in typically used mammalian laboratory animal species).	Studies involving chronic exposures (those involving an extended period of time, or a significant fraction of the subject's lifetime) provide information on potential effects following prolonged and repeated exposure. Ideal dosing regimens include dosing for 5-7 days per week for at least 12 months or greater for chronic studies in rodents. For other species repeated dosing should ideally cover 50 percent or greater of the animal's lifespan.
Chronic criterion (CC)	Highest water concentration of a toxicant or effluent to which organisms can be exposed indefinitely without causing chronic toxicity; CC_{df} - chronic criterion based on protecting humans from exposure to the pollutant from both drinking water and eating sport-caught fish; CC_f - chronic criterion based on protecting humans from exposure to	Highest water concentration of a toxicant or effluent to which organisms can be exposed indefinitely without causing chronic toxicity.			

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
	the pollutant from eating sport-caught fish only; CC_w - a chronic criterion based on protecting wildlife from exposure to the pollutant from eating aquatic organisms				
Chronic standard (CS)	The highest water concentration of a toxicant to which organisms can be exposed indefinitely without causing chronic toxicity	Highest water concentration of a toxicant to which organisms can be exposed indefinitely without causing chronic toxicity. Chronic standards are listed in parts 7050.0222 and 7052.0100.			
Developmental Toxicity				Major manifestations: A. death of developing organism, B. structural abnormality, C. altered growth, and D. functional deficiency.	
Incremental Cancer Risk or Cancer Risk Level	10^{-5}=cancer risk level of one chance in 100,000/ and the incremental cancer risk from exposure to carcinogenic chemicals, singly or in mixtures, shall not exceed one in 100,000		See "Risk Associated Dose"…plausible upper bound incremental cancer risk equal to one in 100,000	See Additional Lifetime Cancer Risk	See Additional Lifetime Cancer Risk
Intake rate (IR)				Rate of inhalation, ingestion, and dermal contact, depending on the route of exposure. For ingestion of water, the intake rate is simply the amount of water, on a per body weight basis, ingested	Drinking water intake (default = 2 L/day for adults) Fish intake (defaults = 0.0175 kg/day for the general population and sport anglers, and 0.142

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
				on a daily basis (liters per kg body weight per day, L/kg-day) for a specified duration. For the derivation of noncancer and cancer HRLs, the time-weighted average of the 95th percentile intake rate for the relevant duration was used	kg/day for subsistence fishers)
Lowest observable adverse effect level (LOAEL)	Lowest tested concentration that caused a statistically significant occurrence of an adverse effect in comparison with a control when all higher test concentrations caused adverse effects		Lowest tested dose or concentration of a substance which resulted in an observed adverse effect in exposed test organisms when all higher doses or concentrations resulted in the same or more severe effects	The lowest exposure level at which a statistically or biologically significant increase in the frequency or severity of adverse effects was observed between the exposed population and its appropriate control group. A LOAEL is expressed as a dose rate in milligrams per kilogram body weight per day (mg/kg-day).	Lowest experimental exposure level at which there are statistically or biologically significant increases in frequency or severity of observed adverse effects between the exposed population and its appropriate control group
Mode of action				The sequence of key event(s) (i.e., toxicokinetics and toxicodynamics) after chemical exposure upon which the toxic outcomes depend.	A MOA is a description of key events and processes starting with the interaction of an agent with a cell, through operational and anatomical changes, and resulting in cancer formation. "Mode" of action is contrasted with "mechanism" of action, which implies a more detailed, molecular description of events than is meant by MOA.

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
Modifying factor (MF)					Determined using professional judgment; provides for additional uncertainty not explicitly included in UF, such as completeness of the overall database and the number of species tested. (The value for MF must be greater than zero and less than or equal to 10; the default value for the MF is 1)
No observable adverse effect level (NOAEL)	Highest tested concentration that did not cause a statistically significant occurrence of an adverse effect in comparison with a control when no lower test concentration caused an injurious or adverse effect		Highest tested dose or concentration of a substance which resulted in no observed adverse effect in exposed test organisms where higher doses or concentrations resulted in an adverse effect	An exposure level at which there was no statistically or biologically significant increase in the frequency or severity of adverse effects between the exposed population and its appropriate control group.	Exposure level at which there are no statistically or biologically significant increases in the frequency or severity of observed adverse effects between the exposed population and its appropriate control; some effects may be produced at this level, but they are not considered as adverse, nor precursors to specific adverse effects
Reference dose (RfD)	Estimate of a daily exposure to the human population, including sensitive subpopulations, that is likely to be without appreciable risk or deleterious effects over a lifetime; expressed in units of daily dose, mg/kg/day			An estimate of a dose for a given duration to the human population, including susceptible subgroups such as infants, that is likely to be without an appreciable risk of adverse effects during a lifetime. It is derived from a suitable dose level at which there is few or no	Traditionally estimated by identifying the most appropriate NOAEL for the critical effect. The LOAEL may be used to estimate the RfD if no appropriate NOAELs have been identified. From IRIS **Reference Dose (RfD):** An estimate (with

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
				statistically or biologically significant increases in the frequency or severity of an adverse effect between the dosed population and its associated control group. The RfD includes one or more divisors, applied to the suitable dose level, accounting for: (i) uncertainty in extrapolating from mammalian laboratory animal data to humans; (ii) variation in toxicological sensitivity among individuals in the human population; (iii) uncertainty in extrapolating from effects observed in a short-term study to effects of long-term exposure; (iv) uncertainty in using a study in which health effects were found at all doses tested; and (v) uncertainty associated with deficiencies in the available data. An HRL is not derived if the product of the divisors exceeds 3,000. The RfD is expressed as mg/kg-day	uncertainty spanning perhaps an order of magnitude) of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. It can be derived from a NOAEL, LOAEL, or benchmark dose, with uncertainty factors generally applied to reflect limitations of the data used. Generally used in EPA's noncancer health assessments. [Durations include acute, short-term, subchronic, and chronic and are defined individually in this glossary].
Reference Dose, Acute					**IRIS:** An estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure for an acute duration (24 hours or less)

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
					to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. It can be derived from a NOAEL, LOAEL, or benchmark dose, with uncertainty factors generally applied to reflect limitations of the data used. Generally used in EPA's noncancer health assessments.
Reference Dose, Short-term					**IRIS:** An estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure for a short-term duration (up to 30 days) to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. It can be derived from a NOAEL, LOAEL, or benchmark dose, with uncertainty factors generally applied to reflect limitations of the data used. Generally used in EPA's noncancer health assessments.

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
Reference Dose, Subchronic					**IRIS:** An estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure for a subchronic duration (up to 10% of average lifespan) to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. It can be derived from a NOAEL, LOAEL, or benchmark dose, with uncertainty factors generally applied to reflect limitations of the data used. Generally used in EPA's noncancer health assessments.
Relative source contribution factor (RSC)	Fraction of the total allowable daily dose of a toxic pollutant that is attributed to drinking water and fish consumption relative to other sources of the pollutant to humans, such as air or food, in the calculation of criteria. In the absence of sufficient data to establish a chemical-specific RSC value, the RSC is 0.2.		Factor (percentage) used in calculating an HNV or HNC to account for all sources of exposure to a contaminant. The RSC reflects the percent of total exposure which can be attributed to surface water through water intake and fish consumption	The percentage (or fraction) of an individual's total permissible exposure to a substance or chemical that is "allocated" to ingestion of water. Application of this factor acknowledges that non-ingestion exposure pathways (e.g., dermal contact with water, inhalation of volatilized chemicals in water) as well as exposure to other media,	A relative source contribution factor is used to account for non-water sources of exposure. (Not used for carcinogens based on a linear low-dose extrapolation) May be either a percentage (multiplied) or amount subtracted, depending on whether multiple criteria are relevant to the chemical.

	MPCA Minn. R. 7050	MPCA Minn. R. 7052	40 CFR 132 1995	MDH Minn. R. 4717	EPA HH-AWQC 2000
				such as air, food, and soil may occur. The Minnesota Groundwater Protection Act, in Minnesota Statutes, section 103H.201, Subd. (1)(d), requires that the Minnesota Department of Health use a relative source contribution in deriving health risk limits for systemic toxicants. MDH determined the following default RSC values: 0.2 for highly volatile contaminants (chemicals with a Henry's Law Constant greater than 1×10-3 atm-m3/mole) and 0.5 for young infants or 0.2 for older infants, children and adults for chemicals that are not highly volatile.	
Short-Term duration				A period of more than 24 hours, up to 30 days.	A short-term exposure study generally refers to multiple or continuous exposure usually occurring over a 14-day to 28-day time period. The purpose of short-term repeated dose studies is to provide information on possible adverse health effects from repeated exposures over a limited time period.

Risk associated dose (RAD)			Dose of a known or presumed carcinogenic substance in (mg/kg)/day which, over a lifetime of exposure, is estimated to be associated with a plausible upper bound incremental cancer risk equal to one in 100,000.		
Risk Specific Dose (RSD)					Based on a linear low-dose extrapolation (mg/kg-day) (dose associated with a target risk, such as $10^{-6})^{-}$
Slope Factor	See Cancer Potency Factor		Also known as q1*, slope factor is the incremental rate of cancer development calculated through use of a linearized multistage model or other appropriate model. It is expressed in (mg/kg/day) of exposure to the chemical in question	An upper-bound estimate of cancer risk per increment of dose that can be used to estimate cancer risk probabilities for different exposure levels. This estimate is generally used only in the lowdose region of the dose-response relationship; that is, for exposures corresponding to risks less than 1 in 100. A slope factor is usually expressed in units of cancer incidence per milligram of chemical per kilogram of body weight per day (per [mg/kg-day] or [mg/kg-day]$^{-1}$).	See Cancer Potency Factor/ IRIS: **Oral Slope Factor:** An upper bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg-day, is generally reserved for use in the low-dose region of the dose-response relationship, that is, for exposures corresponding to risks less than 1 in 100.

Sub-Chronic toxicity (duration)			(appD) adverse effect, measured by assessing an acceptable endpoint, resulting from continual exposure for a period of time less than that deemed necessary for a chronic test.	A period of more than 30 days, up to approximately 10% of the life span in humans (more than 30 days up to approximately 90 days in typically used mammalian laboratory animal species)	Studies involving subchronic exposure (occurring usually over 3 months in animal studies) provide information on health hazards likely to arise from repeated exposure over a limited period of time. Ideal dosing regimens include dosing for 5-7 days per week for 13 weeks or greater (90 days or greater) for subchronic studies in rodents. For other species repeated dosing should ideally cover 10 percent or greater of animal's lifespan.
Toxic Effect				An observable or measurable adverse biological event, or the organ, tissue, or system in which the effect is manifested.	
Toxic Pollutant	References Minn. Stat. 115: means those pollutants, or combinations of pollutants, including disease-causing agents, which after discharge and upon exposure, ingestion, inhalation or assimilation into any organism, either directly from the environment or indirectly by ingestion through food chains, will, on the basis of information available to the			See Developmental Toxicity and Toxic Effect	

	agency (administrator), cause death, disease, behavioral abnormalities, cancer, genetic mutations, physiological malfunctions, including malfunctions in reproduction, or physical deformation, in such organisms or their offspring. (also, 33 USC Section 1362 or 1317(a)(1))				
Uncertainty factor (UF)		One of several numeric factors used in operationally deriving criteria from experimental data to account for the quality or quantity of the available data	One of several numeric factors used in operationally deriving criteria from experimental data to account for the quality or quantity of the available data	One of several factors used in deriving a reference dose from experimental data. UFs are intended to account for: □ the uncertainty in extrapolating from mammalian laboratory animal data to humans, i.e., interspecies uncertainty factor; □ the variation in sensitivity among the members of the human population, i.e., intraspecies variability factor; □ the uncertainty in extrapolating from effects observed in a short-term study to potential effects from a longer exposure, i.e., subchronic-to-chronic uncertainty factor; □ the uncertainty associated with using a study in which health effects were found at all doses tested, i.e., LOAEL-to-NOAEL	Factor which reduces the dose to account for several areas of scientific uncertainty inherent in most toxicity databases. Standard UFs are used to account for variation in sensitivity among humans, extrapolation from animal studies to humans, and extrapolation from less than chronic NOAELs to chronic NOAELs. An additional UF may be employed if a LOAEL is used to define the RfD

				uncertainty factor; and the uncertainty associated with deficiencies in available data, i.e., database uncertainty factor. Uncertainty factors are normally expressed as full or half powers of ten, such as 10^0 (=1), $10^{0.5}$ (=3), and 10^1 (=10). All applicable uncertainty factors are multiplied together to yield a composite uncertainty factor for the RfD. Half-power values such as $10^{0.5}$ are factored as whole numbers when they occur singly but as powers or logs when they occur in tandem (EPA 2002c). Therefore, a composite UF using values of 3 and 10 would be expressed as 30 (3×10^1), whereas a composite UF using values of 3 and 3 would be expressed as 10 ($10^{0.5} \times 10^{0.5} = 10^1$). Uncertainty and variability factors are typically values of three or ten and are multiplied together. The Department has not developed a HRL if the product of all uncertainty factors exceeds 3,000.	

Volatile				Nonvolatile - Henry's Law constant < 3×10-7 atm-m3/mol Low - Henry's Law constant > 3×10-7 to 1×10-5 atm-m3/mol Moderate - Henry's Law constant >1×10-5 to 1×10-3 atm-m3/mol High - Henry's Law constant >1×10-3 atm-m3/mol	
Weight of evidence				An approach requiring a critical evaluation of the entire body of available data for consistency and biological plausibility. Potentially relevant studies should be judged for quality and studies of high quality given much more weight than those of lower quality.	The weight-of-evidence narrative is a summary of the key evidence for carcinogenicity. It describes the agent's MOA, characterizes the conditions of hazard expression including route of exposure and any anticipated disproportionate effects on sensitive subgroups, and recommends appropriate dose-response approach(es). Significant strengths, weaknesses, and uncertainties of contributing evidence are also highlighted.

Headings: EPA HH-AWQC EPA 2000 (b) Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health

AWQC = Ambient Water Quality Criterion (mg/L)
RfD = Reference dose for noncancer effects (mg/kg-day)
POD = Point of departure for carcinogens based on a nonlinear low-dose extrapolation (mg/kg-day), usually a LOAEL, NOAEL, or LED10
UF = Uncertainty Factor for carcinogens based on a nonlinear low-dose extrapolation (unitless)
RSD = Risk-specific dose for carcinogens based on a linear low-dose extrapolation (mg/kg-day) (dose associated with a target risk, such as 10-6)
RSC = Relative source contribution factor to account for non-water sources of exposure. (Not used for linear carcinogens.) May be either a percentage (multiplied) or amount subtracted, depending on whether multiple criteria are relevant to the chemical.
BW = Human body weight (default = 70 kg for adults)
DI = Drinking water intake (default = 2 L/day for adults)
FIi = Fish intake at trophic level (TL) I (I = 2, 3, and 4) (defaults for total intake = 0.0175 kg/day for general adult population and sport anglers, and 0.1424 kg/day for subsistence fishers). Trophic level breakouts for the general adult population and sport anglers are: TL2 = 0.0038 kg/day; TL3 = 0.0080 kg/day; and TL4 = 0.0057 kg/day.
BAFi = Bioaccumulation factor at trophic level I (I=2, 3 and 4), lipid normalized (L/kg)

Table 3-1.: Uncertainty factors and the modifying factor uncertainty factor definition

UF_H Use a 1-, 3-, or 10-fold factor when extrapolating from valid data in studies using long-term exposure to average healthy humans. This factor is intended to account for the variation in sensitivity (intraspecies variation) among the members of the human population.

UF_A Use an additional 1-, 3-, or 10-fold factor when extrapolating from valid results of long-term studies on experimental animals when results of studies of human exposure are not available or are inadequate. This factor is intended to account for the uncertainty involved in extrapolating from animal data to humans (interspecies variation).

UF_S Use an additional 1-, 3-, or 10-fold factor when extrapolating from less-than-chronic results on experimental animals when there are no useful long-term human data. This factor is intended to account for the uncertainty involved in extrapolating from less-than-chronic NOAELs to chronic NOAELs.

UF_L Use an additional 3- or 10-fold factor when deriving a RfD from a LOAEL, instead of a NOAEL. This factor is intended to account for the uncertainty involved in extrapolating from LOAELs to NOAELs.

UF_D Use an additional 1-, 3-, or 10-fold factor when deriving a RfD from an "incomplete" database. Missing studies, e.g., reproductive, are often encountered with chemicals. This factor is meant to account for the inability of any study to consider all toxic endpoints. The intermediate factor of 3 (½ log unit) is often used when there is a single data gap exclusive of chronic data. It is often designated as UF_D.

Modifying factor

Use professional judgment to determine the MF, which is an additional uncertainty factor that is greater than zero and less than or equal to 10. The magnitude of the MF depends upon the professional assessment of scientific uncertainties of the study and database not explicitly treated above (e.g., the number of species tested). The default value for the MF is 1. Note: With each UF or MF assignment, it is recognized that professional scientific judgment must be used.

Appendix A3 — Federal Policies and US EPA Guidance on Improving Risk Assessment Methods for Protection of Infants and Children

Federal Legislation and Orders

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Appendix A4 — Details of the Dissolved and Particulate Organic Carbon Values used in *State* Bioaccumulation Factor Methods

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As stated in the section on Bioaccumulation Factors (BAFs):

> "The goal of EPA's approach for developing BAFs is to represent the long-term average bioaccumulation potential of a pollutant in aquatic organisms that are commonly consumed by humans throughout the United States"(USEPA 2003b).

Part of the BAF methods for nonionic organic pollutants is determining the freely dissolved fraction (f_{fd}) of the pollutant in surface waters. To calculate this, dissolved organic carbon (DOC) and particulate organic carbon (POC) values are needed. For the National BAF Methods EPA compiled data from many ambient surface water types from across the country available in national EPA and United States Geological Survey (USGS) databases. The data were evaluated based on quality assurance and control and other criteria, including omitting data if reporting limits were > 1.0 and 0.2 mg/L for DOC and POC, respectively (a full description is in USEPA 2003, Section 6.3). As analytical methods for DOC, POC, and Total Organic Carbon (TOC) have changed little over time, monitoring data collected between 1980 and 1999 was used to calculate median, national default values of 2.9 mg/L for DOC and 0.5 mg/L for POC. EPA guidance recognizes though that the concentrations of these parameters can vary depending on waterbody type, season, hydrograph, and ecoregions and recommends that states use regional to site-specific data when available (USEPA 2000b, 2003b, 2009).

The DOC and POC values used in the BAF methods do influence the f_{fd} and ultimately the BAF of nonionic organic chemicals with log K_{ow} values greater than 5 as shown in the following table (see further discussion in Section 6.3 USEPA 2003b).

Table A-1: From USEPA 2003, Table 6-13

Table 6-13. Effect of DOC and POC Concentrations on the Freely Dissolved Fraction (f_{fd}) Relative to National Default Values of DOC and POC

Percentile	DOC (mg/L)	POC (mg/L)	Fraction Freely Dissolved (f_{fd}) and [Ratio to National Default]			
			Log K_{ow}			
			5.0	6.0	7.0	8.0
50th (National Default)	2.9	0.5	0.93	0.58	0.12	0.014
5th	0.8	0.06[a]	0.99 [1.1]	0.89 [1.5]	0.44 [3.7]	0.08 [5.5]
10th	1.2	0.09[a]	0.98 [1.1]	0.84 [1.5]	0.35 [2.9]	0.05 [3.8]
25th	2	0.2	0.97 [1.0]	0.74 [1.3]	0.22 [1.8]	0.03 [2.0]
75th	5.4	1.1	0.87 [0.9]	0.40 [0.7]	0.06 [0.5]	0.006 [0.5]
90th	9.7	2.3	0.77 [0.8]	0.25 [0.4]	0.03 [0.25]	0.003 [0.24]
95th	14	3.9	0.67 [0.7]	0.17 [0.3]	0.02 [0.16]	0.002 [0.15]

[a] Estimated value based on statistical parameters from the POC distribution (see text).

EPA's Technical Support Documents for BAF development (USEPA 2003b, 2009) provide guidance for a state to replace the default national DOC and POC values on the basis of defining a *Site* at a scale smaller than the *Nation*. Minnesota has defined two *Sites*: Minnesota's inland lakes and streams (*State* BAFs) and Lake Superior BAFs for human health-based water quality standards (HH-WQSs). As part of the Great Lakes Initiative, the Lake Superior BAF methods already use DOC and POC values of 2 mg/L and 0.04 mg/L, respectively (Minn. R. ch. 7052). These values are much lower than those recommended nationally because of the very low concentration of organic carbon in Lake Superior and other Great Lakes. Those DOC and POC values will remain the basis for BAF development in Minn. R. ch. 7052.

The basis for developing *State* BAFs is available from DOC and POC surface water monitoring data for Minnesota (Table A2). MPCA pulled data from the same sources and years used by EPA when developing national values: USGS National Water Information System (Minnesota NWIS) and Minnesota STORET (STOrage and RETrieval and Environmental Data Access). However, this data search also included more current monitoring data and different evaluation criteria. MPCA identified a narrower suite of datasets that are more representative of average organic carbon parameters across the state's ecoregions and waterbody types. More weight was given to condition monitoring designs over studies that focused on problem investigations.

Table A-2: Available data and sources used to set statewide DOC and POC values

Source of Data	Project Descriptions	Waterbody Type/ Number of Sites	Parameters Measured	Median Values (mg/L)
Minnesota STORET	MPCA Regular Lake Condition Monitoring and Mercury Studies http://www.pca.state.mn.us/water/lake.html	Lakes 46	TOC DOC (estimated as 90% of TOC)	9.1 8.1
Minnesota STORET	MPCA/Minnesota Dept. of Natural Resources Sentinel Lakes Program http://www.pca.state.mn.us/water/sentinel-lakes.html	Lakes 25 (TOC) 24 (DOC)	TOC, DOC (one year) POC (TOC-DOC)	8.5 7.5 0.7
EPA-Minnesota National Lakes Assessment Project	Random survey of 50 Lakes Across Minnesota http://www.pca.state.mn.us/publications/wq-nlap1-02.pdf	Lakes 50	TOC DOC POC (TOC-DOC)	9.2 8.6 0.5
Minnesota STORET	MPCA Milestones and Ambient Trace Metals Condition Monitoring and Trends in Streams Representing all 10 Basins	Streams 108	TOC DOC (estimated as 90% of TOC)	7.9 7.1
USGS NWIS	Condition Monitoring Stream Samples	Streams 186	DOC POC	8.7 1.0

Minnesota STORET at http://www.pca.state.mn.us/water/storet.html
USGS NWIS at http://waterdata.usgs.gov/mn/nwis/qw

The available dataset has strengths in the representative coverage of lakes and streams in the state for TOC and DOC. When reviewing the ranges of measurements and medians, TOC is more variable than DOC. The DOC medians from lakes and rivers ranged from 7.5 to 8.7 mg/L. For the two large datasets that only measured TOC, an estimate of DOC can be made as a percentage of TOC. Evaluations of the Minnesota datasets and by those of other researchers have found the ratios of TOC to DOC to be approximately 0.9 (Wetzel 2001). This provides more information to examine a statewide DOC value.

In contrast, few monitoring activities in the state include analysis for POC—this was also the case in the national dataset. More often TOC is measured or DOC and occasionally both. Measuring TOC and DOC and subtracting to get a POC value is a way to estimate POC; however, as POC is present at lower concentrations than TOC or DOC, subtracting two larger concentrations limits the precision of the resulting POC value. Even with those limitations, the calculated POC medians of 0.5 and 0.7 mg/L from two lake datasets agree with values measured in lakes and reported in EPA's detailed review of DOC and POC for the national defaults (see Section 6.3 in USEPA 2003). In Minnesota, only the USGS streams dataset includes POC analytical results. The median value of 1.0 mg/L is higher than the lakes, but that too is consistent with EPA's findings.

The Minnesota DOC monitoring data on average and as a whole show much higher concentrations than EPA national default median value of 2.9 mg/L. The lowest median values for lakes and streams are actually closer to the 95th and 75th percentiles, respectively, reported for the entire national dataset (see Table A3).

Table A-3: From USEPA 2003b, Table 6-10

	DOC (mg/L)				POC (mg/L)			
Statistic	All Types	Stream/ River	Lake/ Reservoir	Estuary	All Types	Stream/ River	Lake/ Reservoir	Estuary
Median	2.9	3.8	2.1	2.7	0.5	0.6	0.3	0.9
Mean	4.6	5.6	2.9	3.4	1.0	1.3	0.5	1.2
Std.	5.1	5.9	3.0	2.6	2.0	2.5	1.0	1.8
CV	111%	105%	103%	76%	200%	192%	200%	150%
n	111,059	69,589	25,704	15,766	86,540	48,238	23,483	14,819
5th	0.8	0.7	1.0	1.7	0[a]	0[a]	0.08	0.1
10th	1.2	1.0	1.4	2.0	0	0[a]	0.1	0.3
25th	2.0	2.1	1.8	2.3	0.2	0.2	0.2	0.5
75th	5.4	6.9	2.6	3.2	1.1	1.4	0.5	1.4
90th	9.7	11.6	5.0	5.0	2.3	3.1	0.8	2.2
95th	14	16.5	7.8	9	3.9	5	1.3	3
95th/5th	17.5	23.6	7.8	5.3	—	—	16.3	30.0

[a] Values calculated to be less than zero because of measurement error; see Section 6.3.2 for explanation.
Source: U.S. EPA LDC and USGS WATSTORE databases. Data retrieval: January 2000; see Sections 6.3.1 and 6.3.2 for description.

The higher DOC concentrations for Minnesota's lakes and streams are representative of the state's ecoregional physicochemical characteristics. In a supplementary report for EPA's Biotic Ligand Model for Copper, the national DOC dataset for streams and rivers was evaluated by Level III Ecoregions (HydroQual Inc. 2008). The ecoregions in Minnesota—46 to 52[44]— had 25th percentile DOC values (3.9 to 13 mg/L) higher than the national median, supporting the use of higher values for Minnesota *State* BAFs.

For DOC and POC there are not enough data to reliably show large variability in median values between lakes versus streams, or ecoregions, to warrant different *Site* distinctions on a scale other than statewide. Differences are made in terms of other parameters used for BAF development, for example in fish lipid values. There are also procedures described in Minn. R. chs. 7050 and 7052 to modify a WQS based on local, site-specific data in place of the values used to develop HH-WQSs. Given the strengths and limitations of the dataset, more conservative median values will be used for DOC and POC. **A DOC value of 7.5 mg/L reflects the lowest actual median from the datasets and the lowest estimated DOC value from a comprehensive TOC dataset for streams in Minnesota. Given the very limited data for POC, the value applied for *State* BAFs is 0.5 mg/L**; while this concentration equals the lowest

[44] 46: Northern Glaciated Plains, 47: Western Corn Belt Plains, 48: Lake Agassiz Plain, 49: Northern Minnesota Wetlands, 50: Northern Lakes and Forests, 51: North Central Hardwood Forests, and 52: Driftless Area.

estimated median value from the Minnesota datasets, the primary reason to use this value is to retain EPA National default until more monitoring data are available to consider alternate values in a future revision. And for comparison, the table below shows the f_{fd} for a range of Log K_{ow} values using the default EPA national DOC and POC values or Minnesota's.

Table A-4: Comparison freely dissolved fraction (f_{fd}) for different DOC and POC values

	Log Kow			
ffd_{50th}	5	6	7	8
EPA Default	0.93	0.58	0.12	0.01
MN Default	0.90	0.48	0.08	0.01

References

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Appendix A5 — Details on Class 2B (Non-trout) Fish Lipid Values

In 1990 MPCA reviewed available data on fish lipid content to develop State-specific fish lipid values by use classification for application in the human health bioaccumulation factor (BAF) methods (Appendix B3). As part of the adoption of new BAF methods, these lipid values were reviewed for continued use. Described in BAF Methods Section IV.C.g., MPCA is retaining the use of the 8.5% fish lipid value for Lake Superior and 6% for Class 2A, and for Class 2A cold water trout waters. With introduction of the trophic level (TL) distinctions being implemented for statewide use with the proposed revisions, not just for Lake Superior BAFs, MPCA reviewed available data by TL for application to Class 2B (2C, 2D) cool and warm water fisheries.

As part of an interagency Fish Contaminant Monitoring Program (FCMP), fish are collected in Minnesota for routine analysis of mercury and PCB fish tissue concentrations. The Minnesota Department of Agriculture's laboratory analyzes the fish, which has also included fish lipid content. Minnesota data were one of four datasets used in 1990. Regional fish lipid data were also reviewed as part of GLI (USEPA 1995a); while this data came primarily from Great Lakes fish monitoring, this information was included for comparison. EPA's national BAF methods also included examining fish lipid data relative to freshwater and estuarine species commonly consumed (USEPA 2003b). On a national scale, there are many differences in fish species consumed; however, for species that overlap with those found in Minnesota's inland lakes and rivers, those values were included for comparison. In addition, both EPA documents include categorization of fish species into TLs; those categorizations were used to assign TLs to the fish species available in the Minnesota 2010 dataset (Table A5).

Lipid data from the FCMP database as retrieved in March 2010 provide the best available results for examining appropriate cool and warm water fishery (Classes 2B, 2C, 2D) values by TL; this dataset only includes fish collected in Minnesota waters, primarily for the purposes of providing data for Minnesota Department of Health's Fish Consumption Advice. For this reason, the fish data closely represents the target monitoring data for use in evaluating average lipid content in fish eaten in Minnesota. The dataset includes species consumed as fillet samples (both skin on and off) from many inland lakes, streams, and large rivers with over 5,500 results. The available data span the years of 1980 through 2009.

EPA's approach for developing fish lipid values for use in the GLI and national BAF methods was accomplished side-by-side with the review and development of survey data used to recommend default fish consumption rates. For GLI that was the 1991-1992 Michigan surveys by West et al. 1993 and the United States Department of Agriculture's Continuing Survey of Food Intake by Individuals (CSFII 1994-1998) for the national survey. The default TL fish lipid values aimed to represent the consumption-weighted mean percentages by species.

Minnesota's approach in 1990 and for this current review has differed by focusing on the game fish species residing in each surface water classification and using an average fish lipid value representative of those species. This approach does not directly link the fish lipid values with an estimate of actual consumption by species as that data are not sufficiently comprehensive in available regional surveys of sport fishing consumption patterns used to develop Minnesota's

fish consumption rates (Appendix B1). The West et al. 1993 survey data are very comparable to the Wisconsin and Ontario surveys actually used to develop the Minnesota's fish consumption rate of 30 grams per person per day. The use of this survey for GLI is the basis for Minnesota applying the same TL distinctions statewide as those used for the GLI (TL3 24% and TL4 76%); however, the fish lipid data examined centered on fish caught and consumed from the Great Lakes, which can differ from inland lake fish species eaten and fish lipid content. In addition, EPA's attempts to define consumption weighted lipid values also are based on certain assumptions as the survey data are not totally complete to address all aspects of calculating these particular values (USEPA 1995a and Sections 6.2.3-6.2.4 *Uncertainty and Sensitivity Analysis* in USEPA 2003b).

The overall fish lipid data reviewed is available in Table A5; the fish lipid data collected as part of Minnesota's FCMP and downloaded in 2010 provides the most complete and relevant dataset for developing appropriate TL fish lipid values.

Table A-5: Summary of available data on fish lipid content and assigned trophic levels for Minnesota cool-warm water fish species

		Percentages of Mean Fish Lipid Content			
Species	**Trophic Level**	**Minnesota 1990 (Appendix B3)**	**Minnesota 2010**	**GLI Dataset (USEPA 1995a)**	**National Dataset (USEPA 2003)**
Bass, smallmouth	4	1.33	0.8	1.73	
Bass, largemouth	4	0.84	0.5	0.70	
Bass, rock	4	0.39	0.2	0.44	
Bass, white	4	2.62	2.9	4.09	
Bullhead, black	3	1.80	1.2	1.45	
Bullhead, brown	3	1.43	1.1		2.6
Burbot	4	0.85	0.2	0.86	
Carp, common	3	5.07	5.6	8.55	5.4
Catfish, channel	4	4.61	4.4	9.36	5.3
Catfish, flathead	4	1.49	2.0	0.92	
Crappie, mixed	3		0.5		
Crappie, black	3	0.58	0.7		
Crappie, white	3	1.37	0.5		
Drum, freshwater (sheepshead)	(3 assigned based on carp)	3.96	2.0		
Muskellunge	4	1.69	1.1	1.53	1.1
Perch, mixed		0.92			
Perch, yellow	3/4 (GLI >20 cm)	0.78	0.2 (no difference if averaged by size)		1.0
Pike, northern	4	0.71	0.5	1.79	0.6
Sauger	(4- assigned based on walleye)	0.73	0.7		
Sturgeon		2.97			
Sturgeon, lake	4		5.0		9.4
Buffalo, bigmouth (Sucker)	3	4.91	7	8.66	

		Percentages of Mean Fish Lipid Content			
Species	**Trophic Level**	**Minnesota 1990 (Appendix B3)**	**Minnesota 2010**	**GLI Dataset (USEPA 1995a)**	**National Dataset (USEPA 2003)**
Buffalo, smallmouth (Sucker)	3		8.1		
Sucker, redhorse	3	1.99		1.86	
Sucker, Moxostoma Genus (redhorse, golden redhorse, silver redhorse, shorthead redhorse)	3		2.1 (species means 0.4-3.7)		
Sucker, white	3	1.64	1.7	2.03	
Sunfish, bluegill	3	0.73	0.6	0.83	
Sunfish, green	(3 assigned based on bluegill)	0.30			
Sunfish, pumpkin	(3 assigned based on bluegill)	0.73	0.5		
Walleye (Pike)	4	1.10	1.1	1.95	

The Minnesota 2010 species data were further averaged by TL and Genus as shown in Table A6. A few single species lipid data results were not used because of very low sample numbers (N).

Table A-6: Further review and means for final TL values from Minnesota's FCMP database

Species	N	Mean % Lipids Minnesota 2010	Genus Means		Notes
Trophic Level 3					
Bullhead, black	58	1.2	1.2		
Bullhead, brown	12	1.1			
Carp, common	1037	5.6	5.6		
Crappie, mixed	14	0.5	0.7		Only used black crappie (definitive species ID and highest N)
Crappie, black	179	0.7			
Crappie, white	4	0.5			
Drum, freshwater (sheepshead)	134	2.0	2.0		
Perch, yellow	49	0.2	0.2		
Buffalo, bigmouth (Sucker)	36	7	7.6	3.8* Mean of Sucker Genus Used in TL3 Total Mean	
Buffalo, smallmouth (Sucker)	35	8.1			
Sucker, Moxostoma Genus (redhorse, golden redhorse, silver redhorse, shorthead redhorse)	32 (species N: 6-11)	2.1	2.1		

Sucker, white	372	1.7	1.7		
Species	**N**	**Mean % Lipids Minnesota 2010**	**Genus Means**		**Notes**
Sunfish, bluegill	304	0.6	0.6		Bluegill only (higher N)
Sunfish, pumpkin	6	0.5			
		TL 3 Mean	**2.0**		**Mean of 7 Genus***
Trophic Level 4					
Bass, smallmouth	220	0.8	0.7		
Bass, largemouth	151	0.5			
Bass, rock	5	0.2	Not used		Low N
Bass, white	183	2.9	2.9		
Burbot	4	0.2	Not Used		Low N
Catfish, channel	371	4.4	4.4	3.2*	
Catfish, flathead	53	2.0	2.0	Mean of Catfish Genus	
Perch, yellow	49	0.2 (no difference if averaged by size)	0.2		
Muskellunge	11	1.1	0.8		
Pike, northern	990	0.5			
Sauger	93	0.7	0.9		
Walleye (Pike)	1072	1.1			
Sturgeon, lake	9	5.0	5.0		Very limited fishing-only border rivers with one fish per year. (MN DNR)
		TL4 Means	2.0		Includes Sturgeon Mean of 7 Genus*
			1.5		**Without Sturgeon Mean of 6 Genus***

Based on EPA recommendations for BAF development to represent average or mean values, the development of TL fish lipid values to use for cool and warm water fisheries (Classes 2B, 2C, 2D) centered on average or mean values of representative Genus. For TL3 the three "Sucker" Genus means were combined into one to not over represent this family of fish. The two catfish Genus in TL4 were also combined for the same reason. Two different Bass Genus means were retained to account for the large differences in mean lipid values and other differences in Genus characteristics.

MPCA recommended fish lipid values are 2.0% for TL3 and 1.5% for TL4. These lipid values differ only slightly from the 1990 single lipid value of 1.5%. The higher TL3 lipid value is consistent with the higher fat content of benthic, bottom-feeding fish species. The TL3 lipid value is also comparable to the default lipid values developed by EPA for GLI of 1.82% and the

national methods of 2.6%. The TL4 values developed by EPA of 3.1% for GLI and 3.0% for national-use most likely differed from MPCA's values because they included trout and salmon species lipid data, which are not part of MPCA's dataset for Class 2B waters. MPCA uses different lipid values based on trout and salmonids in a single TL application (TL4) for the Class 2A HH-WQS in Lake Superior based on Lake Trout (8.5%) and for other cold water, trout fisheries using another representatively higher lipid value (6%).

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Appendix B1— 1990 Exposure Assumptions – Fish Consumption, Drinking Water, Body Weight and Life Span

The calculation of human health-based aquatic life standards requires the use of several exposure assumptions. Exposure in this context refers to the means or media through which humans are potentially exposed to harmful pollutants. Some of the exposure assumptions have very wide spread use and acceptance among federal and state health and pollution control agencies. Others are less widely used. Values for the latter exposure variables, discussed in the following Section, have been developed by MPCA staff with guidance from the Toxics Technical Advisory Committee (Appendix B-4).

The drinking water consumption value of 2 liters per day used in these equations is near the average value for adults (National Academy of Science 1977). It is the standard value used by the EPA, the Minnesota Department of Health, the World Health Organization, and many states to calculate drinking water standards (e.g., Michigan 1984, Wisconsin 1988). Similarly, the 70 kg (154 pounds) standard weight for an adult is widely used and accepted, as is the 70 year (life time) exposure assumption for carcinogenic chemicals (USEPA 1980, USEPA 1998).

Incidental ingestion of water

The incidental ingestion of small amounts of water by humans during swimming in rivers and lakes not protected as a drinking water source is another potential source of exposure to toxic chemicals. Michigan (1984), for example, assumes the incidental consumption of 0.01 liters of water per day for waters protected for swimming. This value was adopted by MPCA in 1990 and by the Great Lakes Initiative for all the Great Lake States in 1995 (USEPA 1995).

Ten milliliters of water is generally less than one mouth-full of water. While, the ingestion of a small amount of water during swimming almost certainly will not be a daily occurrence over a life time, ingestion may be a common occurrence during the summer months for some children. The ingestion of this amount by swimmers on a daily basis is a reasonable assumption. MPCA staff feels that the inclusion of an incidental water consumption factor is a reasonable safeguard to include in the calculation of human health-based standards. Thus, equations for waters not protected as a source of drinking (Class 2B/2C/2D) waters include the 0.01 liter per day incidental ingestion factor.

Relative source contribution

Humans are exposed to potentially hazardous substances from many sources. Drinking water and the consumption of fish from lakes and streams are just two of the many possible sources. For example, people are exposed to pollutants from the indoor and outdoor air they breathe and the food they eat. Depending on the chemical, the location and dietary habits of the exposed population, the contribution from water and fish, out of the potential total daily exposure, can be relatively small. Some data suggest that drinking water and fish consumption may be a minor source for some pollutants (Olsen 1988, Clark and Fuller 1987). The non-water

and non-fish exposure is accounted for by including a relative source contribution factor (K) in equations 2 and 3.

The relative source contribution factor is applicable to standards for systemic (non-cancer causing) chemicals but it is not used in the calculation of standards for carcinogenic chemicals. The determination of q1*s by the EPA includes conservative steps that build in an extra margin of safety. The risk assumed for standards for carcinogenic chemicals is intended to be well below that which would result in any incremental increased risk to the exposed population. This is consistent with the policies of EPA, MDH, and most states in EPA Region V.

Fish consumption

Sport fishing is very popular in Minnesota. It is estimated that two million of the over four million people living in Minnesota are anglers. Eating the days catch is an important part of the fishing experience for many. Thus, it is appropriate to set standards for toxic substances that will protect that part of the population that catches and eats freshwater fish. A daily fish consumption value is used in the calculation of fish consumption only and drinking water plus fish consumption human health-based aquatic life standards. Since 1980 EPA has used an average fish consumption value of 6.5 grams per day. This figure is based on a diet survey of the general population and includes the consumption of shellfish and estuarine fish as well as freshwater fish (USEPA 1980). A fish consumption figure that is based on the consumption of sport-caught freshwater fish by the angling population, but excludes estuarine and store-bought fish is more appropriate for Minnesota.

Data from several fish consumption surveys were reviewed to arrive at a daily consumption rate. All the surveys show a very large range in the amount of fish consumed by people (Table B-1). A reasonable goal is the protection of 80 percent of the fishing population. This goal will probably protect many anglers in the upper 20th percentile consumption bracket because these people, while eating a lot of fish, may not eat all their fish from the same source over an entire lifetime. An 80th percentile consumption value for the angling population should protect better than 95 percent of the general population (Table B-1). It is impractical and probably overprotective on a statewide basis to extend the level of protection to cover the maximum amount of reported fish consumption. However, MPCA staff may, on a site-specific basis, alter the fish consumption value to protect local populations that may eat lesser or greater amounts of fish from one source over a long period of time.

Minnesota anglers have not been surveyed but data from surveys conducted in Ontario and Wisconsin provide the most applicable data, and these data should adequately represent consumption patterns in Minnesota. Both Wisconsin and Ontario restricted their surveys to their angling populations, and both report the consumption of sport-caught fish (Cox et al. 1985, Cox et al. 1987, and Wisconsin Division of Health 1987).

The fish consumption survey data are summarized in Table B-1. It is apparent that the general population eats relatively little freshwater fish even in states that have large fishing populations (Rupp et al. 1980). In contrast, a survey of people that regularly fish Lake Michigan showed very high fish consumption rates (Humphrey's data in Rupp et al. 1980).

Cox et al. (1987) surveyed Ontario anglers that requested fish consumption advisories. Three separate surveys were conducted (1978, 1983, 1986), and the results were consistent from

survey to survey. Data shown in Table B-1 are averages of the three surveys. The Ontario consumption values are reported as sport-caught fish only.

The Wisconsin Division of Health (1987) surveyed randomly selected persons that had purchased Wisconsin fishing licenses and lived in 10 pre-selected Wisconsin counties. The Wisconsin survey separated sport-caught fish consumption from total fish consumption. Sport-caught fish consumption is about half of the total. Also, Wisconsin's sport-caught fish consumption rates are about half of Ontario's (Table B-1). A more detailed breakdown of the results of the Wisconsin survey is shown in Table B-2.

Estimated 80th percentile sport-caught fish consumption rates are:

56.6 grams/day, Michigan (Lake Michigan)

37.5 grams/day, Ontario

21.0 grams/day, Wisconsin

The mean of the three values is 38.4 grams/day. However, the Wisconsin and Ontario data should be more representative of consumption patterns in Minnesota where the majority of fish caught will be from inland lakes rather than from the Great Lakes. The mean of the Wisconsin and Ontario values is 29 grams/day. This value will be "rounded" to 30 grams/day and will be used to calculate human health-based aquatic life standards.

Table B-1: Summary of fish consumption data from several sources.

Surveyed Population	N	Median	Mean	75%	80%	95%	99%	Source
		Fish Consumption – Grams per Day						
Lake Michigan anglers	182	27.4	45.5	50.1	56.6	103	--	1
General, NW Central[a]	1503	0.0	2.3	--	--	--	25.3	1
General, NE Central[b]	2924	0.0	2.0	--	--	--	25.8	1
Wisconsin anglers (sport-caught fish)	790	6.2	11.3	15.5	21.0	37.3	--	2
Wisconsin anglers (all fish)	797	21.1	25.4	33.6	--	63.4	--	2
Ontario anglers (sport-caught fish)[c]	3020	12.5	20.8	30.7	37.5	105	--	2
		Meal Size – Grams per Meal						
Columbia River anglers	10,900	--	200	--	--	--	--	1
Ontario anglers (sport-caught fish)[c]	2144	227[d]	290	--	357[d]	524[d]	--	3
		Meal Frequency – Meals per Month						
Wisconsin anglers (sport-caught fish)	790	0.8	1.5	2.1	--	--	--	2
Wisconsin anglers (all fish)	797	2.8	3.5	4.5	--	--	--	2
Ontario anglers (sport-caught fish)	1683	1.5	--	4.1[d]	5.0[d]	13.8[d]	--	3

Footnotes:

[a]NW central, includes Minnesota, Iowa, Missouri, N & S Dakota, Nebraska, Kansas

[b]NE central, includes Ohio, Indiana, Illinois, Michigan, Wisconsin.

[c]Consumption values equal meals/day times median meal size of 227 grams.

[d]Data plotted on log probability paper to estimate percentiles.

Sources for Table:
1. Rupp, E., F. Miller and C. Baes. (1980)
2. Wisconsin Division of Health and the State Laboratory of Hygiene. (1987)
3. Cox et al. 1985 and Cox et al. 1987

Table B-2: Fish consumption data from Wisconsin survey. Survey taken in five counties bordering the Great Lakes and in five inland counties.

Statistics	Data Excluding Zeros	Data Including Zeros
Consumption of Sport-caught Fish in Grams per Day		
N	728	790
Median	6.2	6.2
Mean	12.3	11.3
75th %	15.5	15.5
95th %	37.3	37.5
Consumption of All Fish in Grams per Day		
N	786	797
Median	21.8	21.1
Mean	26.1	25.4
75th %	34.2	33.6
Meals per Month – Sport-caught Fish		
N	790	728
Median	0.8	0.8
Mean	1.5	1.6
75th %	2.1	2.1
Meals per Month – All Fish		
N	797	786
Median	2.8	2.9
Mean	3.5	3.5
75th %	4.5	4.5

Source: Ms. Beth Jones, Wisconsin Division of Health, November 3, 1987.

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APPENDIX B2 – 1990 Bioaccumulation Factor Methods in Minn. R. ch. 7050

A. Introduction

The terms bioconcentration and bioaccumulation refer to the process which results in greater concentrations of a chemical in the tissues of aquatic organisms than the concentration of the same chemical in the surrounding water. Bioconcentration refers to uptake of the chemical by aquatic organisms from only the surrounding water. Bioaccumulation refers to uptake of the chemical by aquatic organisms from the surrounding water and all other sources, including the organism's food. For highly lipophilic and persistent chemicals, uptake through the food chain, called biomagnification, substantially increases the concentration of the chemical in aquatic organisms at the top of the food chain. Biomagnification is the stepwise bioaccumulation of a chemical from one trophic level to the next.

The ratio between the concentration of the chemical in an aquatic organism and the surrounding water is the bioaccumulation factor (BAF). A bioconcentration factor (BCF) is the ratio between the concentration of the chemical in an aquatic organism and the surrounding water when the organism is exposed to the chemical only via the water (see definitions in Appendix A). BAFs are best measured in the field and BCFs are usually measured in laboratory experiments. For chemicals that are only moderately bioaccumulative (BAF less than about 1000, at 7.6 percent lipid), the BAF and BCF are nearly equal (i.e., little or no biomagnification). For highly bioaccumulative chemicals, the BAF for an organism may be a factor of 10 or more greater than the BCF. For this reason, BAFs are preferred over BCFs for the calculation of standards.

Most bioaccumulative organic chemicals have a strong affinity for fatty tissues (lipid) in aquatic organisms. Thus, it is important to know the lipid content of the organism, particularly fish, to determine the BAF because the BAF will vary directly with lipid content. Many BAFs or BCFs taken from the literature are based on the tissue of the whole test organism. Whole fish BAFs or BCFs can be adjusted to edible portion BAFs or BCFs if the percent lipid has been measured in the test organisms.

Final BAFs are determined through three main procedures, which are described in order of their preference.

- The preferred procedure is to use field measured BAFs and measured percent lipid values, when these data are available. Field measured BAFs without lipid data can be used as is.
- The second procedure is the use of laboratory measured BCFs and a food chain multiplier, with or without associated percent lipid data.
- The third and least preferred procedure is used when no measured BAFs or BCFs are available and the BAF must be predicted based on the chemical's octanol/water partition coefficient (K_{ow}).

B. Determination of bioaccumulation factors

1. The applicable BAF is determined when field measured BAFs and (preferably) percent lipid data are available as follows:
   a. Field measured steady-state BAFs are assembled from the literature, EPA criteria documents and other sources, and reviewed for acceptability (Stephan et al. 1985). The BAFs may be for the edible portion or whole body of the test fish. The percent lipid in the tissue, edible portion or whole organism, of the test organisms should be reported as part of the study.
   b. Each acceptable BAF is normalized to one percent lipid by dividing the BAF by the arithmetic average percent lipid for the test organism used. If the percent lipid is not reported for the actual test organisms, data from the same population of test organisms may be used; otherwise the BAFs usually cannot be normalized with lipid data from other information sources because of the variability in lipid levels even in the same species of fish. The BAF can be determined without lipid adjustment.
   c. The geometric mean of all the normalized BAFs for each species is calculated.
   d. The final normalized BAF is the geometric mean of all the acceptable field measured species mean normalized BAFs.
   e. The final BAF for a chemical is adjusted for percent lipid as follows:
      1) Cold water fish (Class 2A): the normalized mean BAF is multiplied by 6 percent lipid.
      2) Cool and warm water fish (Class 2B, 2C, and 2D): the normalized mean BAF is multiplied by 1.5 percent lipid. See Appendix F for an explanation of the percent lipid values used.
2. If field measured BAFs are not available, the applicable BAF is determined by applying a food chain multiplier to a measured BCF. BCFs are adjusted for percent lipid as in the case of BAFs.
   a. Laboratory measured steady-state BCFs are assembled from the literature, EPA criteria documents and other sources, reviewed for acceptability, and normalized to one percent lipid, as described for BAFs. A BCF can be determined without lipid adjustment if lipid data are not available.
   b. The geometric mean of all the normalized BCFs for each species is calculated.
   c. The final normalized BCF is the geometric mean of all the acceptable species mean normalized BCFs.
   d. The final normalized BAF is determined by multiplying the BCF by the appropriate food chain multiplier (FCM) from equation 7 or 8. Table B-3 shows example FCMs for a range of log K_{ow} and parachor values (see Section VII.C.). If parachor data for the chemical is not available, the food FCM can be based on log K_{ow} alone (equation 8). Examples of the food chain multipliers used in Minn. R. ch. 7052 (GLI rule) are shown in Table B-7.
   e. The final BAF for a chemical is adjusted for the mean percent lipid as follows:

1) Cold water fish (Class 2A): the normalized mean BAF is multiplied by 6 percent lipid.
2) Cool and warm water fish (Class 2B, 2C, and 2D): the normalized mean BAF is multiplied by 1.5 percent lipid.

3. If neither field measured BAFs nor laboratory measured BCFs are available for a lipophilic chemical, a BAF can be predicted based on the chemical's log K_{ow}, and the application of a FCM. Estimated BAFs are adjusted for percent lipid. This method is not applicable to inorganic chemicals.
   a. A BCF can be predicted based on the relationship between BCFs and the octanol/water partition coefficient (K_{ow}). The relationship demonstrated by Veith and Kosian (1983), as expressed by equation 6, is used to predict a BCF.

      $\log_{10} BCF = 0.79 \log_{10} K_{ow} - 0.40$ (at 7.6 % lipid) (6)

      Where: log K_{ow} = the log to base 10 of the octanol/water partition coefficient.

The model used to predict a BCF in Minn. R. ch. 7052 is equation 14, Section IX. If measured log K_{ow} values are not available in the literature, they may be estimated using quantitative structure activity relationships (QSAR).

   b. The predicted BCF is converted to a predicted BAF by multiplying the BCF by the appropriate FCM from equation 7 or 8 (Table B-3). If parachor data for the chemical is not available, the FCM can be based on log K_{ow} alone.
   c. The percent lipid of the organisms used to establish the BCF to log K_{ow} relationship is 7.6 (Stephan et al. 1985). Predicted BAFs are adjusted for percent lipid as follows:
      1) Cold water fish: the predicted BAF is normalized to one percent lipid by dividing the BAF by 7.6, and then adjusted for edible portions by multiplying by 6.0 percent lipid.
      2) Cool and warm water fish: the predicted BAF is normalized to one percent lipid by dividing the BAF by 7.6, and then adjusted for edible portions by multiplying by 1.5 percent lipid.

C. Food chain multipliers

As mentioned above, the more bioaccumulative a chemical is, the less BCF data are likely to be representative of the true bioaccumulation potential in nature. The literature supports using a factor to adjust BCFs to better estimate BAFs for chemicals with a log K_{ow} in the range of 3 to 6. The MPCA reviewed this literature and through regression analysis determined the relationship between the log of the ratio of measured BAFs to BCFs, the log K_{ow}, and the chemical's parachor (MPCA 1989). The parachor[45] is an index of the physical properties of the molecule that affect how readily a chemical bioaccumulates. The food chain multiplier (FCM) is determined using equation 7, or equation 8 if parachor information is not available.

[45] Parachor - The surface tension adjusted molar volume. More specifically, parachor is the molecular weight of a liquid times the fourth root of its surface tension, divided by the difference between the density of the liquid and the density of the vapor in equilibrium with it; essentially constant over wide ranges of temperature. Parachor relates to the physical properties of molecule that affect its potential to bioaccumulate in aquatic organisms.

$$Log_{10}\ FCM = 0.384 \log K_{ow} - 0.00055\ Parachor - 1.128 \quad (7)$$

A log k_{ow} of 6 is used to determine the FCM for chemicals that have log K_{ow} values greater than 6.0. If parachor data are not available, that part of the formula can be left out altogether and the formula becomes:

$$Log_{10}\ FCM = 0.384 \log K_{ow} - 1.128. \quad (8)$$

Factors calculated using equation 7 or 8 will not be less than one nor greater than 15 (see Table B-3).

Table B-3: Food chain multipliers for various Log K_{ow} values.

Log K_{ow}	FCM (at parachor of 500)	FCM (no parachor data)
3.0	0.6 (1.0 is used)	1.1
3.5	0.9 (1.0 is used)	1.6
4.0	1.4	2.6
4.5	2.1	4.0
5.0	3.3	6.2
5.5	5.1	9.6
6.0 and greater	8.0	15

4. Examples of BAF determinations.

Table B-4: Example 1: available data for chemical with log K_{ow} of 4.5 and a parachor of 500, no measured lipid data.

BAF/BCF Data	Species	% lipid	Species mean BAF/BCF	Mean BAF/BCF Not lipid normalized
Measured BAF:				Vertebrate
2500	Fathead minnow	NA	2500	2500
Measured BCF:				Invertebrates
900	Daphnia magna	NA	1100	878
1350	Daphnia magna	NA		
700	Mollusk sp.	NA	700	
BAF = BCF x FCM of 2.1	NA	NA	NA	1844
BAF, Log K_{ow} predicted	NA	NA	NA	3001

NA = Not available

In example 1 there is not a large body of BAF or BCF data available, and there is no associated lipid value for the BAF or any of the BCFs.

- Field measured BAF. Since field measured BAFs are preferred, the single BAF of 2500 would be selected as the applicable final BAF in this example.
- BAF from lab measured BCFs times a FCM. The available BCFs are for invertebrate species and, in general, fish BAFs or BCFs are much preferred over invertebrate data. The food chain multiplier (FCM) for a chemical with a log K_{ow} of 4.5 from equation 7 is 2.1. The mean BCF times the FCM of 2.1 equals 1844.

- Predicted BAF from log K_{ow}. The predicted non-lipid-normalized BAF in example 1 equals 3001 (BCF from equation 6 equals 1429; times the FCM of 2.1 equals 3001 at 7.6 % lipid). The BAFs determined by the three methods, measured BAF, BCF times a FCM, and predicted BAF, are all reasonably close. This gives support to the lone measured BAF, which is used as the final BAF. However, this example illustrates the uncertainty introduced into BAF determinations when percent lipid data are not available. Fathead minnows tend to have a higher fat content (~8%) than is assumed for warm water game fish (1.5%). Thus, the BAF of 2500 may be on the high side for use with warm water fisheries. Also, if the predicted BAF is adjusted for the lipid value assumed for equation 6 (7.6%), the predicted BAF becomes 592 (3001 (1.5/7.6)) for warm water fisheries and 2369 (3001 (1.5/7.6) for cold water. Also, if the predicted BAF is adjusted for the lipid value assumed for equation 6 (7.6%), the predicted BAF becomes 592 (3001 (1.5/7.6)) for warm water fisheries and 2369 (3001 (6.0/7.6)) for cold water fisheries.

Table B-5: Example 2: available data for chemical with log K_{ow} of 5.3 and a parachor of 500, lipid data available.

BAF/BCF Data	Species	% lipid	Species mean BAF/BCF Normalized to 1% lipid	Mean BAF/BCF At 1.5% lipid
Measured BAF:				
None				
Measured BCF:				Vertebrate
4000	Rainbow trout	7.9	691	1193
5000	Rainbow trout	5.3		
7500	Fathead minnow	8.2	915	
BAF = BCF x FCM of 4.3	NA	NA	NA	5130
BAF, Log K_{ow} predicted	NA	NA	NA	5197

In example 2 there is no measured BAF but there is percent lipid data available for the BCFs. Therefore, the final BAF is based on the mean BCF times a FCM.

- Field measured BAF. None.
- BAF from lab measured BCFs times a FCM. The adjusted BAF of 5130 for warm water fisheries (20,511 for cold water fisheries), is the geometric mean of the two species mean BCFs times the FCM of 4.3 from equation 7. This BAF would be used as the applicable final BAF in this example since no field measured BAFs are available, and it is based on three measured BCFs.
- Predicted BAF from log K_{ow}. The predicted BAF of 5197 (BCF from equation 6 of 1209 times 4.3 from equation 7) is in agreement with the FCM adjusted BCF of 5130.

D. Historical discussion on differences between methods in Minn. R. chs. 7050/7052

The main differences between the EPA's GLI and Minn. R. ch. 7050 BAF methods is that the EPA methods: 1) provides four methods to determine BAFs rather than three (see Table B-6), 2) uses a different food-chain multiplier model to estimate a BAF from BCF data, and 3) uses a different model to estimate a BCF from K_{ow}. An important similarity is that both methods list, as the preferred BAF method, the use of field measured, lipid adjusted, BAFs. Generally, the GLI BAF methods are more sophisticated and require more data than the Minn. R. ch. 7050 methods. For example, the GLI method varies the BAF with ambient levels of particulate organic carbon and dissolved organic carbon, although default values can be used in the absence of measured data.

Table B-6: Comparison of BAF hierarchy

Preference of method	Minn. R. ch. 7052. GLI	Minn. R. ch. 7050
1	Field measured, lipid adjusted, BAF	Field measured, lipid adjusted, BAF
2	BSAF, BAF estimated from concentration of chemical in sediment	No such method
3	BAF estimated from lab measured BCF and FCM	BAF estimated from lab measured BCF and FCM
4	BAF estimated from K_{ow} and FCM	BAF estimated from Log K_{ow} and FCM

For the first differences, the EPA's GLI methods introduced a means to determine a BAF from concentrations of a chemical in bottom sediments. This method, the Biota-sediment accumulation factor (BSAF) method, is the second preferred of the four GLI BAF methods, and it is not included in Minn. R. ch. 7050. The full method is described in USEPA (1995a). The BSAF is defined as the ratio, in kg of organic carbon to kg of lipid, of a substance's lipid-normalized concentration in the tissue of an aquatic organism to its organic carbon-normalized concentration in surface sediment. Acceptable situations for use of the BSAF are where the ratio does not change substantially over time, both the organisms and its food are exposed, and the surface sediment is representative of average conditions in the vicinity of the organism (See USEPA 1995a and 1995b). A reference measured BAF is needed to use the BSAF method.

The second difference is in the models used to estimate food chain multipliers. The food chain multiplier is used to determine a BAF from laboratory measured BCFs. The EPA model, developed by Gobas, is a more sophisticated model than the one developed by MPCA staff and adopted into Minn. R. pt. 7050.0218 in 1990 (see Table B-7). Specifically, the EPA model accounts for biomagnification, unlike the MPCA's methods that do not.

The third difference is the model used to estimate a BCF from K_{ow}. The EPA's GLI model is very simplistic; K_{ow} equals the BCF at 100 percent lipid. Thus, a predicted BAF is determined by:

$$\text{BAF} = (\text{FCM})\ (\text{predicted BCF}) = (K_{ow})\ (\text{FCM}) \quad \text{at 100 \% lipid} \qquad (14)$$

The Minn. R. ch. 7050 model is:

BAF = (FCM) (predicted BCF)

Where:
predicted log BCF = 0.79 log K_{ow} – 0.40 (6)
and the predicted BCF = $10^{\log BCF}$
at 7.6 % lipid

The lipid normalized (1 % lipid) BAFs from the two models are reasonably close at Log K_{ow} values of 5 and less but very different for the very highly lipophilic chemicals. Table B-7 compares the predicted BAFs and the Food Chain Multipliers from the two methods for a range of log K_{ow} values.

Table B-7

Log K_{ow}	Minn. R. ch. 7052 (GLI)			Minn. R. ch. 7050		
	BCF	FCM	BAF	BCF	FCM	BAF
3	10	1.007	10	12.28	1.0	12
4	100	1.072	107	75.72	1.4	106
5	1000	2.612	2612	466.9	3.3	1541
6	10,000	15.996	159,960	2879	8.0	23,032
7	100,000	26.242	2,624,200	2879	8.0	23,032*
8	1,000,000	7.798	7,798,000	2879	8.0	23.032*

*Both the predicted BCF and FCM models cap out at a log K_{ow} of 6.0

References

Stephan, C.E., D.J. Mount, D.J. Hansen, J.H. Gentile, G.A. Chapman, and W.A. Brungs. 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratories, Duluth, MN; Naragansett, RI, Corvalis, OR. 98 p. (This document is undated but was made available in 1985.)

USEPA. 1995a. Great Lakes water quality initiative technical support document for the procedure to determine bioaccumulation factors. EPA 820-B-95-005. March 1995. Washington D.C.

USEPA. 1995b. Final water quality guidance for the Great Lakes system. Federal Register 60:15366-15425, March 23, 1995.

APPENDIX B3 —Percent Lipid Data for Human Health Bioaccumulation Factors

The bioaccumulation factor (BAF) is an important element in the calculation of human health-based aquatic life standards, particularly if the chemical is moderately to highly bioaccumulative. BAFs are obtained from the literature or estimated from physicochemical properties of the chemical. Bioconcentration factors take into account uptake of substances from the water across external membranes. It is recognized that aquatic organisms can also accumulate substances from their diet and from bottom sediments. Bioaccumulation, which refers to the uptake of substances from all sources, is a better indicator of what occurs in nature. Unfortunately, bioaccumulation factors are not available for many substances (Stephan et al. 1985). Most BAFs and BCFs are obtained from the following sources:

1. EPA criteria documents. EPA reviewed the bioaccumulation information on the chemicals for which they publish criteria. These BAFs and BCFs are used by the MPCA staff when available.
2. AQUIRE computerized data base of toxicity information.
3. Scientific literature. Measured steady state BAFs and BCFs from laboratory or field studies. Reported factors will be normalized to one percent lipid if the lipid content of the test organism was reported.
4. Estimates of BAFs based on the quantitative structure activity relationships (QSAR) of chemicals.

Because of the affinity of lipophilic organic chemicals to fat tissues the percent lipid of the test organism should be reported for the edible portion or the whole fish along with the BAF/BCF. The BAF/BCF is normalized to one percent lipid by dividing the BAF/BCF by the reported mean percent lipid. The percent lipid in muscle tissue is usually less than the percent lipid in the whole organism.

The normalized BAF/BCF is multiplied by 1.5 percent lipid to calculate a BAF for cool and warm water fisheries, and by 6 percent lipid to calculate a BAF for cold water fisheries. The default values of 1.5 and 6 percent should be used unless there is adequate site-specific data available to support a different value. Eight and a half percent lipid is used to calculate standards applicable to Lake Superior under Minn. R. pt. 7052.0110.

The default percent lipid values are based on the lipid data from four sources. The sources are the lipid data collected as part of the Wisconsin (Wisconsin 1988) and Minnesota fish contaminants programs, the U.S. Department of Agriculture (1987), and Bowes and Church (1980). When more than one value was available for one species, an average of the values was used. Data for 33 species of cool and warm water fish and 12 species of cold water fish were available (Table F-1).

Consistent with the recommendations of the Toxics Technical Advisory Committee, central values are used from the lipid data for the default values. The median value of 1.5 percent lipid for cool and warm water species of fish was selected in place of the mean value of 2.5 because most of the high percent lipid values are for nongame fish (median percent lipid in nongame fish = 2.17; median percent lipid in game fish = 0.88). Anglers typically do not eat nongame species.

If evidence shows that nongame fish do make up a substantial portion of a population's diet, a site-specific percent lipid value based on that information can be used to determine a local standard.

The higher mean value was used as the default value for cold water fish because essentially all the salmonids are considered game fish.

References

Bowes and H. N. Church. Revised by Pennington, J.A.T. and H.N. Church. 1980.

Food values of portions commonly used. 14th Ed. Harper and Row, N.Y. 257 p.

Stephan, C. E., D. J. Mount, D. J Hansen, J. H. Gentile, G. A. Chapman, and W. A. Brungs. 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratories, Duluth, Minnesota; Naragansett, RI, Corvalis, OR. 98 p. (This document is undated but was made available in 1985).

U.S. Department of Agriculture. 1987. Consumption of foods: finfish and shellfish products - raw -processed -prepared. U.S.D.A. Agriculture Handbook Number 8-15.

Wisconsin Department of Natural Resources. 1988. Technical support document for Chapter NR 105 of the Wisconsin Administrative Code. Wisconsin DNR. Unpublished document. 49 p.

Table F-1. LIPID CONTENT OF FISH, EDIBLE PORTIONS

Fish Species	Type	Percent Lipid Values					Mean Values		
		MN	Bowes & Church	USDA	WI DNR	WI DNR L.Superior	All species	Non-sal. species	Salmonid species
Bass, smallmouth	G	0.53	2.6		0.86		1.33	1.33	
Bass, largemouth	G	0.72			0.96		0.84	0.84	
Bass, rock	G				0.39		0.39	0.39	
Bass, white	G	2.82	2.3		2.74		2.62	2.62	
Buffalofish	R	5.61	4.2				4.91	4.91	
Bullhead, black	R	1.8					1.80	1.80	
Bullhead, brown	R	1.77			1.09		1.43	1.43	
Bullhead, yellow	R	0.75			1.37		1.06	1.06	
Burbot	R		0.9	0.8			0.85	0.85	
Carp	R	5.11	4.2	5.6	5.37		5.07	5.07	
Catfish, channel	G	5.4	3.1	4.26	5.68		4.61	4.61	
Catfish, flathead	G	1.29			1.68		1.49	1.49	
Cisco	LS	2.1		1.91			2.01		2.01
Crappie, white	G		0.8		1.94		1.37	1.37	
Crappie, black	G	0.5			0.66		0.58	0.58	
Drum, freshwater	G	1.79	5.2	4.9			3.96	3.96	
Muskellunge	G				1.69		1.69	1.69	
Perch, mixed	G			0.92			0.92	0.92	
Perch, yellow	G	0.75	0.9		0.7		0.78	0.78	
Pike, northern	G	0.5	1.1	0.69	0.53		0.71	0.71	
pike walleye	G	0.85	1.2	1.22	1.13		1.10	1.10	
Quilback	R	1.6					1.60	1.60	
Redhorse, shorthead	R	2.34					2.34	2.34	
Redhorse, silver	R	1.5	2.3				1.90	1.90	
Salmon, atlantic	LS		13.4	6.34			9.87		9.87
Salmon, chinook	LS	2.5	15.6	10.44		2.53	7.77		7.77
Salmon, coho	LS					3.29	3.29		3.29
Salmon, pink	LS		3.7	3.45			3.58		3.58
Sauger	G	0.73					0.73	0.73	
Shad, sp unkown	R			13.77			13.77	13.77	
Shad, gizzard	R		14				14.00	14.00	
Seepshead	R		2.8	2.41			2.61	2.61	
Smelt	LS		6.2	2.42		2.78	3.80		3.80
Sturgen	G		1.9	4.04			2.97	2.97	
Sucker, Redhorse	R	1.99					1.99	1.99	
Sucker, white	R	1.16	1.8	2.32	1.27		1.64	1.64	
Sunfish, bluegill	G	0.64			0.82		0.73	0.73	
Sunfish, green	G				0.3		0.30	0.30	
Sunfish, pumpkinseed	G			0.7	0.76		0.73	0.73	
Trout, brook	T	1.7	2.1		5	4.48	3.32		3.32
Trout, brown	T	2.24			4.19	9.41	5.28		5.28
Trout, lake	T	8.33	37.1		7.8	11.1	16.08		16.08
Trout, rainbow	T		11.4	3.36	0.8	5.72	5.32		5.32
Trout, mixed	T			6.61			6.61		6.61
Whitefish, mixed	LS		8.2	5.86	0.82		4.96		4.96
				SUMMARY		Mean	3.44	2.51	5.99
						Median	1.99	1.49	4.96

% Lipid Data From:
Minnesota Pollution Control Agency
Bowes and Church 1980
USDA 1987
Wisconsin DNR 1988
Wisconsin DNR for Great Lakes fish

Type: G = game fish; R = rough fish; LS = Lake Supreior fish; t = trout

APPENDIX B4–Toxics Technical Advisory Committees

The first of these advisory committees was formed by the MPCA in 1988 specifically to review the procedures used to determine aquatic life standards. The stated purpose of this committee, called the Toxics Technical Advisory Committee (TTAC), was *to advise the MPCA staff on the technical aspects of developing water quality standards to protect aquatic organisms from the harmful effects of toxic substances so that the procedures will be technically sound and defensible*. The committee reviewed EPA and MPCA procedures used to establish criteria and standards based on the toxicity of chemicals to aquatic life, the protection of human consumers of fish, and the protection of wildlife consumers of aquatic organisms. The committee met monthly for 11 months. They reviewed many issues and recommended a position on most of them (Toxics Technical Advisory Committee 1989). These guidelines reflect the adoption of nearly all their recommendations, including a list of the TTAC members.

The TTAC made many recommendations to the MPCA which were incorporated into the methods adopted into Minn. R. ch. 7050 in 1990. Notable recommendations are:

- Assume a 30 g/day fish consumption amount
- Use BAFs rather than BCFs
- Use a relative source contribution factor
- Assume an incidental consumption of water for swimmers
- Use a Tier II method to calculate toxicity-based standards when data are scarce.

In March 1996, at the request of potentially affected parties, the MPCA formed an advisory committee to review GLI issues. This committee met five times and recommended the following items that are relevant to this guidance.

- Recommended that Minnesota use the state-wide fish consumption rate of 30 grams per day in the Lake Superior basin in place of 15 grams per day used by the GLI
- Recommended that Minnesota use the same percent lipid values for BAFs used state-wide (6 % for trout waters and 1.5 % for non-trout waters) in place of the percent lipid values used by the GLI
- Recommended that Minnesota adopt a separate percent lipid value for Lake Superior of 8.5 percent.

In the fall of 1996 the MPCA formed another citizen advisory committee, the Water Quality Standards Advisory Committee (WQSAC). The WQSAC met monthly over a 15 month period. Two additional meetings were held specifically to discuss the new draft EPA ammonia criterion. The main focus of this committee was aspects of MPCA procedures used to set water quality-based effluent limits, but the WQSAC reviewed the following water quality standard issues that are relevant to this guidance.

- Recommend dissolved metal standards in place of total metal standards
- Recommend labeling existing standards in Minn. R. ch. 7050 as to whether they are toxicity- human health- or wildlife-based

- Reviewed the current averaging periods (duration) for standards (no action taken)
- Reviewed the current compliance frequencies for standards (no action taken)
- Recommend the adoption of the GLI Tier II method
- Choose not to review the fish consumption amount of 30 grams per day thereby accepting that value

Table C-1: The TTAC members are listed below with their affiliations at the time the committee was in session, 1988-1989.

Direct Toxicity Subcommittee	
Mr. Paul Aasen Metropolitan Waste Control Commission	Mr. Mohamed Elnabarawy Minnesota Mining and Manufacturing
Dr. Ira Adelman, Head Department of Fisheries and Wildlife University of Minnesota	Dr. Steven Hedtke, Chief EPA Monticello Ecological Research Station
Mr. Craig Anderson Minnesota Power	Mr. David Lane City of St. Cloud League of Minnesota Cities
Mr. David Bonistall, Supervisor Environmental and Analytical Services Champion International Corp.	Dr. Paul Toren, President Izaak Walton League of Minnesota
Mr. Jay Unwin National Council of the Paper Industry	
Human Health Subcommittee	
Mr. Douglas Alms City of Red Wing Minnesota Wastewater Operators Assoc.	Dr. Jeff Stevens School of Public Health University of Minnesota
Mr. Bruce Anthony Koch Refining Co.	Mr. Eric Swanson Citizens for a Better Environment
Mr. David Gray Environmental Health Division Minnesota Department of Health	Dr. Robert Keiger Delta Environmental Consultants, Inc.
Mr. Glen Kuhl, Supervisor Environmental Studies Northern States Power	Mr. Jack Skrypek, Chief Ecological Services Minnesota Department of Natural Resources
Mr. Joe Stepun Western Lake Superior Sanitary District	
Wildlife Subcommittee	
Mr. Calvin Blanchard Pesticide Regulatory Section Department of Agriculture	Dr. Gerald Niemi, Associate Director Center for Water and Environment Natural Resources Research Institute
Dr. Terry Kreeger Department of Fisheries and Wildlife University of Minnesota	Mr. Stan Smith U.S. Fish and Wildlife Service
Mr. Charles Hoffman Pickands Mather and Company	Mr. Clarence Trushenski Minnesota Conservation Federation
Mr. Roger Murnane Nico Products	Mr. Robert Glazer Carlos Avery Research Minnesota Department of Natural Resources

Members at Large	
Mr. George Crocker North American Water Office	Dr. Jeffrey Foran National Wildlife Federation
Dr. James Luey EPA Region 5	Mr. Mark Deutschman HDR Engineering, Inc.
MPCA Staff	
Dr. David Maschwitz	Ms. Carolyn Dindorf
Dr. Velma Charles-Shannon	Mr. Curtis Sparks
Mr. Raymond Reyes	

APPENDIX C — US EPA Organoleptic (Taste and Odor) Criteria

EPA organoleptic (taste and odor) criteria (USEPA 1998b) were adopted unchanged as Chronic Standards (CS) in Minn. R. ch. 7050 when the concentration was lower than the toxicity-based, human health-based, or wildlife-based CS. The table below lists the EPA taste and odor criteria.

Table C1: EPA organoleptic (taste and odor) criteria

Substance	Criterion mg/L	Substance	Criterion mg/L
Acenaphthene	20*	2-Methyl-4-chlorophenol	1800
Monochlorobenzene	20*	3-Methyl-4-chlorophenol	3000
2,4,6-Trichlorophenol	2*	3-Methyl-6-chlorophenol	20
4-Chlorophenol	0.1	2-Chlorophenol	0.1
3-Chlorophenol	0.1	2,4-Dicholophenol	0.3
2,3-Dichlorophenol	0.04	2,4-Dimethylphenol	400
2,5-Dichlorophenol	0.5	Hexachlorocyclopentadiene	1.0
2,6-Dichlorophenol	0.2	Nitrobenzene	30
3,4-Dichlorophenol	0.3	Phenol	300
2,3,4,6-Trichlorophenol	1.0	Copper	1000
Pentachlorophenol	30	Zinc	5000
2,4,5-Tricholorophenol	1.0		

* Adopted as the MPCA standard.

Appendix D1—Class 2 Human Health-based Water Quality Standards: Current and Proposed for Revision

Note: Revised CSs will be adopted in future rulemakings.

NUMERIC WATER QUALITY STANDARDS FOR MINNESOTA CLASS 2 WATERS						
Human Health-Based Chronic Standards or Criteria Values (less stringent than CS-Tox) Only#						
MINNESOTA POLLUTION CONTROL AGENCY						
All units are in micrograms per liter (µg/L) unless noted otherwise.						
CHEMICAL		**STANDARDS:**				
	Lake Superior Class 2A CS	**Class 2A CS**	**Class 2Bd CS**	**Class 2B, 2C, 2D CS**	**Basis For CS**	**Algorithms 1990 or revised**
Metals and Elements						
Aluminum, total		Not Developed	Not Developed	Not Developed	NA	
Antimony		5.5	5.5	280	PCA Hs	1990
Arsenic, total	2†	2.0	2.0	53	PCA Hs	1990
Cadmium, total		2.23	2.23	4.53	PCA Hs	1990
Chromium III, total		9119	9119	14364	PCA Hs	1990
Chromium VI, total		66	66	146	PCA Hs	1990
Cobalt		2.8	2.8	140	PCA Hs	1990
Copper, total		Not Developed	Not Developed	Not Developed	NA	
Lead, total		Not Developed	Not Developed	Not Developed	NA	
Mercury, Elemental total	0.00153	0.0069	0.0069	0.0069	EPA Hs M	1990
Mercury, Methyl Fish Tissue- 2008 (mg/kg)		0.2	0.2	0.2	EPA Hs	NA
Nickel, total		297	297	15050	PCA Hs	1990
Selenium, total		906	906	2290	PCA Hs	1990
Silver		19.8	19.8	350.0	PCA Hs	1990
Thallium		0.28	0.28	0.56	PCA Hs	1990
Zinc, total		Not Developed	Not Developed	Not Developed	NA	1990
Organic Pollutants						
Acenaphthene		20	20	20	EPA Ho	NA
Acetochlor		85	85	213	PCA Hs	1990
Acrylonitrile (c)		0.38	0.38	0.89	PCA Hc	1990
Alachlor (c)		3.8	4.2	103	PCA Hc	1990
Anthracene		119	407	505	PCA Hs	1990
Atrazine		3.4	3.4	100	PCA Hc	1990
Benzene (c)	10 (5.3†)	5.1	6	98	PCA Hc	1990
Bromoform		33	41	466	PCA Hc	1990
Carbon Tetrachloride (c)		1.9	1.9	5.9	PCA Hc	1990
Chlordane (c)	0.000040	0.000073	0.00029	0.00029	EPA Hc M	1990
Chlorobenzene (Monochlorobenzene)		20	20	20	EPA Ho	NA
Chloroform (c)	278	53	53	603.9	PCA Hc	1990
Chlorpyrifos		4.9	4.9	6.4	PCA Hs	1990
Dichlorodiphenyltrichloroethane (I	0.000025	0.00011	0.0017	0.0017	EPA Hc M	1990
1,2-Dichloroethane (c)		3.5	3.8	190	PCA Hc	1990
Dieldrin (c)	0.0000012	0.0000065	0.000026	0.000026	EPA Hc M	1990

Di-2-ethylhexyl phthlate (c) (bis--)(DEHP)		1.9	1.9	2.1	PCA Hc	1990
Di-n-octyl phthalate		Not Developed	Not Developed	Not Developed	NA	
Endosulfan		0.0076	0.029	0.031	PCA Hs	1990
Endrin	0.0039†	0.0039	0.016	0.016	EPA Hs M	1990
Ethylbenzene		228	454	1360	PCA Hs	1990
Fluoranthene		7.1	26.4	29.2	PCA Hs	1990
Heptachlor (c)		0.00010	0.00039	0.00039	EPA Hc M	1990
Heptachlor Epoxide (c)		0.00012	0.00048	0.00048	PCA Hc	1990
Hexachlorobenzene (c)	74	0.000061	0.00024	0.00024	PCA Hc	1990
Lindane (BHC-gamma) (c)	0.08	0.0087	0.032	0.036	EPA Hc M	1990
Methylene Chloride (c)	46	45	46	1940	PCA Hc	1990
Metolachlor		300	525	2064	PCA Hs	1990
Naphthalene		65	81	81	PCA Hs	1990
Parathion		Not Developed	Not Developed	Not Developed	NA	
Pentachlorophenol (PCP)	0.93†	0.93	1.9	5.5	PCA Hc	1990
Phenanthrene		Not Developed	Not Developed	Not Developed	NA	
Phenol		4080	4080	4080	PCA Hs	1990
Polychlorinated biphenyls (PCB),	0.0000045	0.000014	0.000029	0.000029	EPA Hc M	1990
1,1,2,2-Tetrachloroethane (c)		1.1	1.5	13.0	PCA Hc	1990
Tetrachloroethylene (c)		3.8	3.8	8.9	PCA Hc	1990
Toluene	3725	1380	2050	13100	PCA Hs	1990
Toxaphene (c)	0.000011	0.00031	0.0013	0.0013	EPA Hc M	1990
1,1,1-Trichloroethane		Not Developed	Not Developed	Not Developed	NA	1990
1,1,2-Trichloroethylene (c)	22	25	25	120	PCA Hc	1990
2,4,6-Trichlorophenol		2.0	2.0	2.0	EPA Ho	NA
Vinyl Chloride (c)		0.17	0.18	9.2	PCA Hc	1990
Xylene (total m,p and o)		11600	11600	65100	PCA Hs	1990
Minn. R. ch. 7052 only						
Cyanide	596				EPA Hs	1990 (1998 BAF)
2,4,-Dimethylphenol	368				EPA Hs	1990 (1998 BAF)
2,4,-Dinitrophenol	53				EPA Hs	1990 (1998 BAF)
Hexachloroethane (c)	1.0				EPA Hc	1990 (1998 BAF)
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (TCDD) (c) (pg/L)	0.0014				EPA Hc	1990 (1998 BAF)
†Derived by reference to Minn. R. ch. 7050						
#Disclaimer						
The water quality standards listed in this table are a subset of all the standards applicable to waters of the state; consult						
Minn. Rules Chapter 7050 or 7052 for the complete and official listing of all standards applicable to waters of the state.						
See Minn. Rules Part 7050.0222 for fecal coliform, radioactive materials and temperature standards .						
Footnotes, Explanation of Terms and Abbreviations						
CS = Chronic Standard						
Class 2A = Cold water fisheries and aquatic community (supports salmonids); also designated for drinking water						
Class 2Bd = Cool or warm water fisheries and aquatic community; also protected as a source of drinking water						
Class 2B/2C = Cool or warm water fisheries and aquatic community; NOT protected as a source of drinking water						
Class 2D = Wetlands and associated aquatic community						
(c) = Chemical identified in 1990-2008 as human carcinogen						
(NLC) = Updated 2012 methods recognize Nonlinear Carcinogens						

Appendix D2—Human Health-based Site Specific Water Quality Criteria (2007-2012)

NUMERIC WATER QUALITY SITE-SPECIFIC CRITERIA FOR MINNESOTA CLASS 2 WATERS							
	Lake Superior Class 2A	**Class 2A**	**Class 2Bd**	**Class 2B, 2C, 2D**	**Basis For CC**	**Algorithms 1990 or Revised**	
Perfluorooctanic Acid (PFOA)							
Lake Calhoun, Aug. 2007	NA	NA	NA	1.62 µg/L	PCA Hs	1990	
Miss. R.-Pool 3, Aug. 2007	NA	NA	NA	2.7 µg/L	PCA Hs	1990	
Perfluorooctane Sulfonic Acid (PFOS)							
Lake Calhoun, Rev. May 2010	NA	NA	NA	6.1 ng/L	PCA Hs	1990	
Miss. R.-Pool 2, Rev. Oct. 2009	NA	NA	NA	7 ng/L	PCA Hs	1990	
http://www.pca.state.mn.us/index.php/waste/waste-and-cleanup/cleanup-programs-and-topics/topics/perfluorochemicals-pfc/perfluorochemical-pfc-waste-sites.html							
2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD)	see individual permits				EPA Hc M	1990	
2,3,7,8-Tetrachlorodibenzo-p-Furan (TCDF)					EPA Hc M	1990	
And Penta to Octa Congeners Toxic Equivalency Factors							
Basis for Site-specific Effluent Limits-Values based on Minn. R. ch. 7052							
Basis Codes							
EPA = Standard based on Environmental Protection Agency criterion							
PCA = Standard developed by Minnesota Pollution Control Agency staff							
Hc = Human health-based standard; standard based on cancer effects to humans							
Hs = Human health-based standard; standard based on non-cancer (systemic) effects to humans							
Ho = Standard based on EPA organoleptic criterion (chemical imparts unacceptable taste or odor to fish tissue)							
M = standard based on EPA criterion, modified by MPCA staff							
Other = Standard based on some other end point							