Iowa Wasteload Allocation (WLA) Procedure

Iowa Department of Natural Resources
Water Quality Bureau

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Table of Contents

1.0 Introduction

The purpose of this document is to provide the technical methodologies and procedures used to develop wasteload allocations and water quality-based effluent limits for pollutants of concern that are necessary for the protection of surface water quality standards as described in 567 IAC Chapter 61 – Water Quality Standards. A Wasteload Allocation (WLA) is the portion of a receiving water's total assimilative capacity that is allocated to one of its existing or future point sources of pollution.

The procedures contained in this document will be used as default procedures unless scientifically defensible, site-specific information is provided to justify, to the department's approval, an alternative evaluation to address any issue addressed in this document.

The department develops WLAs for facilities that discharge treated or untreated wastewater (e.g., domestic sewage treatment plant discharges and cooling water) into Waters of the State in order to assure the permitted effluent limits meet applicable state Water Quality Standards. The calculation of a WLA is based on conservative assumptions to protect the water quality under worst-case scenario conditions. Facilities may choose to submit site-specific information on both the receiving waterbody and the discharge characteristics for consideration. If site-specific data are unavailable, then state default values are used.

The WLAs are calculated to protect all downstream uses. A WLA is performed for the protection of each downstream beneficial use and the most stringent WLA governs the final water quality based limits.

The WLA Procedure document is divided into sections to describe the input parameters that impact the wasteload allocation calculations such as discharge flows, critical stream low flows, and the stream flow velocities to name a few. This document presents the wasteload allocation procedures for parameters including ammonia nitrogen, metals, total residual chlorine, chloride, sulfate, bacteria, temperature, and pH. The implementation of general water quality criteria, site-specific data collection requirements, and mixing zone procedures are also established. The WLA Procedure document establishes procedures for biochemical oxygen demand (BOD_5) and dissolved oxygen (DO) modeling. This modeling is used to derive effluent limits for: BOD_5 or carbonaceous biochemical oxygen demand ($CBOD_5$), ammonia-nitrogen (NH_3-N), and dissolved oxygen.

The WLA Procedure document establishes the permit derivation procedure that is used to translate a WLA into a water quality based NPDES permit limit. The WLA Procedure document also includes antidegradation implementation requirements and alternative site-specific methodologies for water quality based limits. The following Sections describe each topic in detail.

The WLA Procedure document addresses the water quality based effluent limits for point source discharges requiring a National Pollutant Discharge Elimination System (NPDES) permit. 567 IAC 64.4(1) *"e"* specifically states that an NPDES permit is not required for any introduction of pollutants from non-point source agricultural and silviculture activities, including storm water runoff from orchards, cultivated crops, pastures, range lands, and forest lands. As a result, the WLA Procedure is not applicable to those discharges as defined in 567 IAC 64.4(1) *"e"*.

2.0 Discharge Flow Determination

WLA calculations require the discharge flow from a point source facility. For continuous wastewater discharges requiring construction permits, WLA analyses are performed for the projected design Average Dry Weather (ADW) and Average Wet Weather (AWW) wastewater discharge flows entering a receiving stream.

Design flows for wastewater treatment plants are obtained from facility plans, engineering reports, or construction permits. Controlled discharge lagoons (CDLs) are designed to have a minimum detention time of 180 days and discharge twice per year. Thus, WLAs for controlled discharge lagoons are calculated using the drawdown rate, which is expected to be ten times the 180 day AWW design flow. The definitions for ADW and 30 day AWW flows for continuous discharges are provided in Section 14.4.5.1 of the Iowa Wastewater Facilities Design Standards and 567 IAC 60.2. For CDLs, the definition of the 180 day AWW design flow is provided in Section 18C.4.1.1 of the Iowa Wastewater Facilities Design Standards.

For industrial discharges with no wastewater design flows, the ADW and AWW flows are the maximum monthly average discharge flow (highest 30-day average discharge flow) and the daily maximum discharge flow, respectively. The maximum monthly average flow and daily maximum flow are defined as follows:

> ***Maximum Monthly Average Flow:*** The Monthly Average Flow is the sum of the total daily discharges by volume during a calendar month, divided by the total number of days during the month that measurements were made. The highest 30-day average discharge flow is the highest value for all the calculated 30-day average discharge flows in the reporting period.
>
> ***Daily Maximum Flow***: The highest total discharge by volume measured during a 24-hour period in the reporting period.

3.0 Design Stream Low Flow Determination

Which Design Stream Low Flow regime to use in the wasteload allocation process depends on the applicable numeric criteria. Aquatic life criteria are derived based on different time durations. For example, the acute criterion duration for ammonia nitrogen represents a one-hour average, and the chronic criterion duration represents a 30-day rolling average. The human health criteria for carcinogens are derived assuming a lifetime exposure. EPA guidance (U.S. EPA [1], 1985 and 304(a) criteria documents) states that criteria are to be protective for an average frequency of excursion of every three years. It is important to note that numeric criteria consist of three components: magnitude, duration, and frequency. Duration and frequency are reflected in the design stream low flow used to develop WLAs. Table 3.0-1 shows the applicable design low flows (also referred to as critical low flows) for the implementation of different criteria in WLAs.

Table 3.0-1. *Design Stream Low Flow Regime*

Numeric Criteria	Design Low Flow Regime
Aquatic Life Protection (TOXICS)	
Acute	1Q10[a]
Chronic	7Q10[b]
Aquatic Life Protection (AMMONIA – N)	
Acute	1Q10
Chronic	30Q10[c]
Human Health Protection & MCL[1]	
Non-carcinogenic	30Q5[d]
Carcinogenic	Harmonic mean Flow[2]
CBOD[3]	7Q10

[a] 1Q10 = 1-day, 10-year low flow

[b] 7Q10 = 7-day, 10-year low flow

[c] 30Q10 = 30-day, 10-year low flow

[d] 30Q5 = 30-day, 5-year low flow

[1] MCL = Maximum Contaminant Level

[2] Harmonic Mean Flow is the number of daily flow measurements divided by the sum of the reciprocals of those daily flows.

[3] CBOD = Carbonaceous Biochemical Oxygen Demand

The Design Stream Low Flow Regime shown in Table 3.0-1 will be used as the default stream flows for calculating wasteload allocations. For point source discharges driven by rainfall events such as combined sewer overflow (CSO) and storm water discharges, water quality based limits may be calculated using other alternative minimum stream flows when those flows are scientifically defensible and when the resulting wasteload allocations are shown to be protective of the water quality standards for the receiving waterbodies.

3.1. Design Stream Low Flows at USGS-Gaged Locations

Annual design stream low flows at the United States Geological Survey (USGS)-gage locations are obtained from the USGS using USGS-approved statistical methods and USGS-approved flow records. The USGS periodically updates annual design stream low flow values. The most recent USGS published stream low flows for USGS gages are used for WLA calculations.

Facilities also have the option to submit site-specific critical low flow data to the department for consideration when new flow records become available. Site-specific critical low flows are calculated using DFLOW or any other USGS approved statistical methods such as SWSTAT and Log-Pearson Type III models.

3.2. Design Low Stream Flows at Ungaged Locations

Design stream low flows on ungaged stream reaches are determined using the most recent low flow-regression equations developed by USGS for Iowa streams. The most current information regarding low flows is presented in the USGS report, *Methods for Estimating Selected Low-Flow Frequency Statistics and Harmonic Mean Flows for Streams in Iowa, Scientific Investigation Report 2012-5171 (revised in May 2013).* This publication will be referred to as the “USGS low-flow study” for the rest of this document.

The USGS may update stream low flow methodology for ungaged locations in the future. The most up-to-date USGS published low stream flow methodology for ungaged locations will be used to derive wasteload allocations when they are applicable.

3.3. Monthly or Seasonal Low Stream Flow Estimation

The USGS low-flow study provides annual or seasonal low-flow regression equations. In certain situations, monthly or seasonal critical low flows in lieu of annual critical low flows are used in order to develop monthly limits. For example, monthly limits are calculated for ammonia nitrogen and temperature and are calculated using monthly critical stream low flows on a case-by-case basis. Department staff developed an Excel program that uses Log-Pearson Type III method and has pre-calculated the monthly critical low flows for all USGS gages using the most recent flow records that have more than 10 years of USGS approved flow data. The department explores the use of monthly critical low flows on a case-by-case basis and considers factors including, but not limited to, the following:

(1) Whether the receiving stream is a perennial stream;

(2) Whether the receiving stream is an effluent-dominated stream; and

(3) Whether there is a nearby USGS gage that has an adequate flow record that can be used to reasonably estimate the monthly low flows at the discharge location.

In addition to the use of critical low flows, an alternative stepwise flow approach is used if a facility chooses to not discharge below a specified stream flow and only discharges when stream flow is high enough to assimilate the discharge. The water quality based limits will be derived using the specified stream flow in lieu of the critical low flows. This approach is used if the facility clearly demonstrates that there is sufficient storage available to operate in this manner and has accurate means of determining stream flow at the point of discharge.

4.0 Wasteload Allocations for Ammonia Nitrogen and Toxics

Wasteload allocations are calculated for a facility using the discharge flows and stream flows as defined in the Discharge Flow Determination and Design Stream Low Flow Determination sections.

The ADW discharge flow is used to derive chronic and acute concentration WLAs; the AWW discharge flow is used to derive chronic and acute mass loading WLAs, as shown in Table 4.0-1.

Table 4.0-1. *Discharge Flows Used to Derive WLAs*

Wasteload Allocations	Discharge Flow* (mgd)
Chronic and Acute Concentration (mg/L)	ADW
Chronic and Acute Mass Loading (lbs/day)	AWW

*: See Discharge Flow Determination Section for definitions

If the facility demonstrates the discharge flow is positively correlated with stream flow, that is, the AWW discharge flow does not occur at critical low flow stream conditions, then the mass loadings are derived based on the concentration WLAs and the AWW discharge flow. The following sections describe the ammonia and toxics WLAs in more detail.

5.0 Ammonia Nitrogen

The aquatic life criteria for ammonia nitrogen is a function of pH for acute criteria and pH and temperature for chronic criteria as presented from Table 3a to Table 3c of 567 IAC 61.3(3) “b”. This is due to the influence of temperature and pH on the toxic form of ammonia (unionized). Therefore, statewide default in-stream and effluent pH and temperature values are established and used when applying the acute and chronic ammonia criteria. The ammonia criteria are calculated based on statewide default monthly pH and/or temperature values. As a result, ammonia WLAs are expressed as monthly values.

Facilities have the option to submit site-specific in-stream and effluent pH and temperature data that can be used in the WLA calculations in lieu of using statewide default values. The data are used to develop site-specific water quality based effluent ammonia nitrogen limits for the facility if the site-specific data are applicable. For more information on site-specific data collection, refer to the Site-Specific Data Collection Section of this document.

5.1. Ambient Background pH, Temperature and Ammonia Nitrogen

The receiving waterbody background pH, temperature, and ammonia nitrogen levels are calculated using the available monitoring data. The available monitoring data are divided into different categories based on the waterbody characteristics and designated uses such as the Mississippi River, the Missouri River, Warmwater, Coldwater and Lakes. The default statewide ambient background levels will be updated periodically when new ambient monitoring data become available. The updated ambient background levels will be used to derive wasteload allocations for ammonia nitrogen.

5.2. Statewide Effluent pH and Temperature as related to Ammonia Nitrogen

Statewide effluent pH and temperature values are derived for different treatment types based on available effluent monitoring data. The effluent pH and temperature values will be updated periodically when new monitoring data become available. The updated effluent pH and temperature values will be used to derive wasteload allocations for ammonia nitrogen.

5.3. Ammonia Nitrogen Decay Calculations

Ammonia nitrogen is non-conservative in the environment and can be oxidized to nitrite and nitrate. Ammonia nitrogen can also be taken up by algae. This ammonia decay is accounted for in a WLA when effluent flows through a conveyance such as discharge pipe, storm sewer, tile line, or a general use stream prior to entering a designated use stream. When sufficient site-specific field data are available, the ammonia nitrogen decay in a general use segment of a stream may be determined by water quality modeling programs, such as QUALIIK. In the absence of sufficient site-specific field data, the department will use a first-order decay equation with a default decay rate coefficient (based on the default value described in Carbonaceous Biological Oxygen Demand ($CBOD_5$) and Dissolved Oxygen (DO) WLAs Section).

In the presence of nitrifying bacteria, ammonia is oxidized first to nitrite, then to nitrate. The chemical reaction is given in Equation 5.3-1:

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \tag{5.3-1}$$

The oxidation reaction is first-order and has the form shown in Equation 5.3-2.

$$N_a = N_{a0}e^{-K_N*t} \tag{5.3-2}$$

Where:

N_a = ammonia nitrogen remaining at any time, t, mg/L

N_{a0} = initial ammonia nitrogen concentration, mg/L

K_N = nitrification rate

t = time, days

The nitrification rate (K_N) is dependent on temperature. The rate is adjusted depending on the effluent temperature as shown in Equation 5.3-3.

$$K_N = K_{N,0}\ 1.083^{(T-20)} \qquad (5.3\text{-}3)$$

Where:

K_N = temperature adjusted nitrification rate
$K_{N,0}$ = nitrification rate, 0.3/day at 20°C
T = effluent temperature, °C

The above default nitrification rate is used to estimate ammonia nitrogen decay unless other scientifically defensible site-specific decay rates are available.

5.4. Mixing Zone and Zone of Initial Dilution as related to Ammonia Nitrogen

The allowed Mixing Zone (MZ) stream flow and the Zone of Initial Dilution (ZID) stream flow for ammonia nitrogen are a function of the dilution ratio of the receiving stream. The allowable stream flow in the MZ and the ZID are defined in 567 IAC 61.2(4) "e" for a specific facility as being dependent on the ratio of the critical stream low flow to the effluent design flow. The chronic and acute wasteload allocations for ammonia are calculated based on the 30Q10 stream flow for chronic WLAs and the 1Q10 stream flow for acute WLAs. The dilution ratio for ammonia nitrogen WLA is calculated as the ratio of stream 7Q10 flow to the effluent discharge flow.

The flow used in the ammonia nitrogen WLA calculations for the MZ and ZID vary with the dilution ratio. The facility is separated into one of three categories based on the river and discharge flows as derived from 567 IAC 61.2(4) "e":

(1) The dilution ratio of stream flow to discharge flow is less than or equal to 2:1 – MZ is 100% of the 30Q10 and ZID is 5% of the 1Q10

(2) The dilution ratio of stream flow to discharge flow is less than or equal to 5:1 and greater than 2:1 – MZ is 50% of the 30Q10 and ZID is 5% of the 1Q10

(3) The dilution ratio of stream flow to discharge flow is greater than 5:1 – MZ is 25% of the 30Q10 and ZID is 2.5% of the 1Q10

Facilities have the option to submit site-specific mixing zone study data, either from field studies or modeling such as the use of the CORMIX model, to the department for consideration in lieu of the above default MZ and ZID values. MZ study data are collected based on the procedures described in Mixing Zone Procedures Section of this document.

MZ boundary pH and temperature values used to meet chronic criteria are defaulted to the ambient statewide background values unless site-specific or regional ambient background pH and temperature values are available (567 IAC 61.2(4) “f”). ZID boundary pH and temperatures are calculated based on the following mass balance equations when site-specific alkalinity and pH data are not available:

$$ZID\ pH = - LOG \{(Q_e * 10^{-pHe} + Q_r * ZID * 10^{-pHr})/(Q_e + Q_r * ZID)\} \qquad (5.4\text{-}1)$$

$$ZID\ TEMPERATURE = [(Q_e * T_e) + (Q_r * ZID * T_r)]/(Q_e + Q_r * ZID) \qquad (5.4\text{-}2)$$

Where:
Q_e = Effluent flow, (cfs)
Q_r = Stream low flow, 1Q10 is used as the Q_r (cfs)
pH_e = Effluent pH, standard unit
pH_r = Upstream flow pH, standard unit
ZID = Zone of Initial Dilution, dimensionless, between 0-1

Facilities have the option to submit site-specific effluent and stream background pH and alkalinity for consideration. If the site-specific data are acceptable, the site-specific pH value at the boundary of the ZID is calculated using the procedure in Appendix A.

5.5. Calculation of the Wasteload Allocations for Ammonia Nitrogen

To meet the acute and chronic aquatic life criteria, the ammonia nitrogen wasteload allocations are calculated based on Equation 5.5-1:

$$(C_r * Q_r * MZ\ or\ ZID) + C_e * Q_e = C_s(Q_r * MZ\ or\ ZID + Q_e) \qquad (5.5\text{-}1)$$

Where:
C_r = Receiving waterbody ammonia nitrogen background concentration, mg/L
Q_r = Design Stream Low Flow (30Q10 for chronic WLA, 1Q10 for acute WLA)
Q_e = Design Effluent flow, cfs (ADW for concentration limits, AWW for mass limits)
Qr*(MZ or ZID) = Stream low flow in the MZ or ZID, cfs
Qr*(MZ or ZID)+Q_e = Total flow in the MZ or ZID, cfs
C_e = WLA concentration (or allowed discharge concentration), mg/L
C_s = Applicable water quality standard, mg/L (acute or chronic criteria)

The equation is solved four times for C_e, one time each for ADW acute, ADW chronic, AWW acute, and AWW chronic. This results in WLAs that are protective of the acute and chronic criteria.

The acute WLAs calculated from Equation 5.5-1 are compared to the allowable ammonia nitrogen concentrations needed to meet the dissolved oxygen standard that are determined according to the water quality modeling procedures described in the $CBOD_5$ and DO WLAs Section. The most stringent acute WLA governs the final ammonia nitrogen limits. The final acute and chronic WLAs for ammonia nitrogen are then carried forward to the Permit Derivation Procedure to determine the water quality-based effluent limits. The final concentration limits are derived using the ADW design flow and the mass limits are derived using the AWW design flow.

6.0 Toxics (Metals and Other Parameters)

This section describes the wasteload allocation calculations for pollutants shown in Table 1 of 567 IAC 61.3(3) including metals, pesticide chemicals, and other toxics.

6.1. In-Stream Background Chemical Concentrations

Iowa Water Quality Standards have defined numerical criteria for different chemicals. To properly implement these criteria and calculate WLAs for each point source discharge, background concentrations of the pollutants in Iowa surface waters are determined based on available ambient monitoring data. The background chemical concentrations will be updated periodically when new ambient monitoring data become available.

Facilities have the option to submit site-specific stream background chemical concentrations for consideration in lieu of the statewide default background concentrations. For more information on site-specific data collection, refer to Site-Specific Data Collection section of this document.

6.2. Mixing Zone and Zone of Initial Dilution for Toxics

The regulatory MZ and ZID are included in 567 IAC 61.2(4) “b”. Each facility has the option of providing site-specific MZ data either by field study or through modeling using programs such as the CORMIX model. Department staff uses the default MZ values as defined in 567 IAC 61.2(4) “b” unless an applicant provides applicable site-specific mixing zone data. Other models, in addition to CORMIX, are used where appropriate or when they become available.

The Mixing Zone Procedure section presents the basic MZ study field data procedures that an applicant provides for recalculation of a local MZ. The purpose of the MZ study is to more closely approximate the local MZ using site-specific data instead of default MZ values, as defined at 567 IAC 61.2(4) “b”(1).

6.3. Calculation of WLAs for Toxics

The calculation of toxic WLAs involves the regulatory MZ and ZID for each point source discharge, the effluent flow rates, and the applicable acute and chronic water quality criteria.

As noted in 567 IAC 61.2(4), the chronic criteria shall be met at the boundary of the MZ; and the acute criteria shall be met at the boundary of the ZID. A mass balance of pollutants shown in Equation 6.3-1 is used to calculate the effluent limits necessary in order to meet these boundary conditions.

$$(C_r * Q_r * \text{ MZ or ZID}) + C_e * Q_e = C_s(Q_r * \text{MZ or ZID} + Q_e) \qquad (6.3\text{-}1)$$

Where:
C_r = Receiving Waterbody Toxics Background concentration, mg/L
Q_r = Design Stream Low Flow (7Q10 for chronic WLA, 1Q10 for acute WLA)
Q_e = Design Effluent flow, cfs (ADW for concentration limits, AWW for mass limits)
Qr*(MZ or ZID) = Stream low flow in the MZ or ZID, cfs
Qr*(MZ or ZID)+Q_e = Total flow in the MZ or ZID, cfs
C_e = WLA concentration (or allowed discharge concentration), mg/L
C_s = Applicable water quality standard, mg/L (acute or chronic criteria)

This equation is solved four times for C_e, one time for ADW acute, ADW chronic, AWW acute, and AWW chronic. This results in WLAs for the protection of the acute criteria as well as wasteload allocations for the protection of the chronic criteria. These wasteload allocation values are then carried forward to the Permit Derivation Procedure section. The final concentration limits are derived using the ADW design flow and the mass limits are derived using the AWW design flow.

7.0 Total Residual Chlorine (TRC)

Point source discharge facilities may use chlorine to disinfect the effluent to meet bacteria limits. Total Residual Chlorine (TRC) effluent limits are calculated for any point sources discharging TRC into or impacting one of the five Class B waters or general use waters to protect aquatic life. The applicable stream numeric criteria are listed in 567 IAC Chapter 61. Alternative disinfectants such as bromine, paracetic acid, etc. may also be used. For alternative disinfectants that do not currently have numerical criteria, the water quality based limits are derived based on the Narrative Water Quality Standards section.

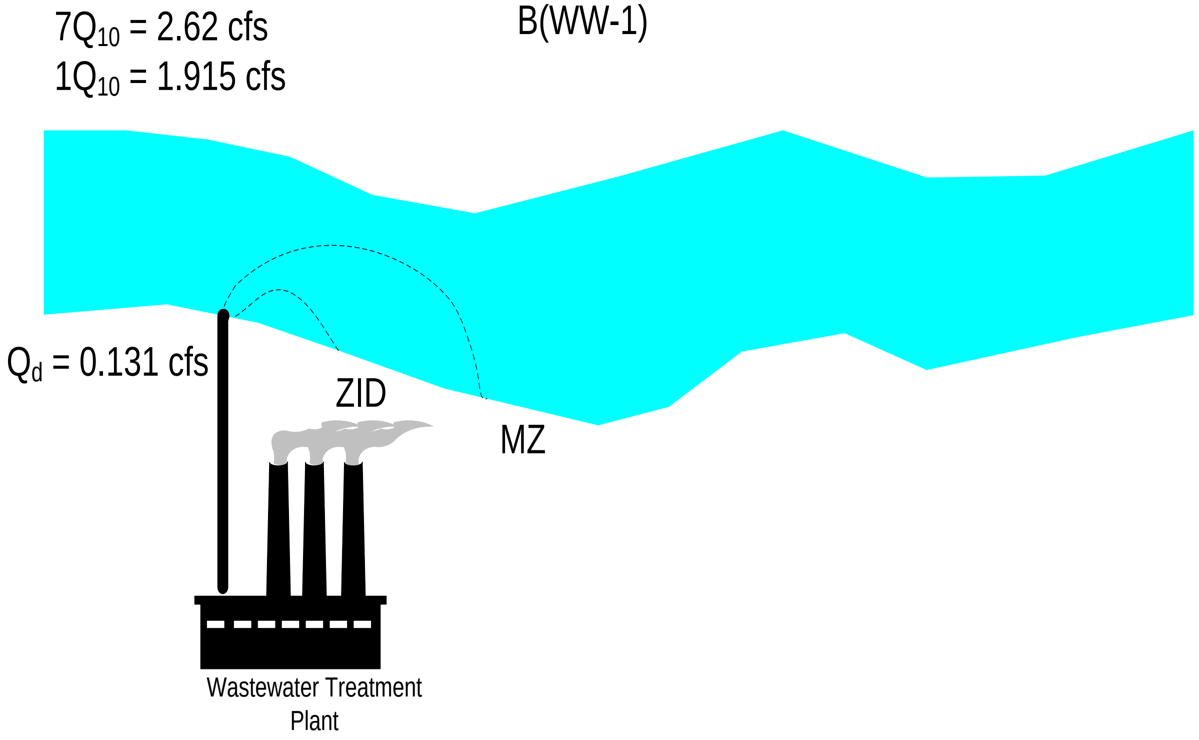
7.1. TRC Wasteload Allocation Calculations

TRC WLAs may be calculated using a combination of mass balance calculations and a first-order decay of TRC. TRC decay is determined based on a first-order equation with a default decay rate of 20/day (U.S. EPA[1], 1984). The TRC decay equation is used to calculate TRC decay in a general use reach, discharge pipe, storm sewer or tile line. Background flow, defined as the sum of all upstream flows and any incremental flows along the modeled reach, can be added at one of the reach entries in the TRC decay calculation. The incremental flows are included at the appropriate distance below the discharge.

Studies have shown that a significant loss of TRC in waters receiving chlorinated wastewater discharges is due to immediate chlorine demand. This demand is due to the highly reactive nature of chlorine with organic matter and bacteria present in the receiving water (U.S. EPA, 1984[1]; Douglas D. and George R. Hetz, 1985). In addition to the TRC decay calculations for a general reach, facilities have the option to collect site-specific TRC decay data in the mixing zone and the zone of initial dilution and submit the data to the department for consideration.

Two sets of example calculations are shown for TRC: one for a direct discharge to a Class B(WW-1) designated use waterbody and the other for discharge to a general use reach that then flows into a Class B(WW-1) designated use waterbody. The background TRC concentration is considered as zero due to the reactive nature of TRC in ambient conditions.

Example 1: Direct Discharge to a Class B(WW-1) Designated Water



The $WLA_{chronic}$ and WLA_{acute} values are calculated using the TRC Mass Balance equation for the designated portion of the receiving stream. The WLA calculations by default do not consider the immediate TRC demand exerted by the receiving stream within the MZ and ZID. Facilities have the option to collect site-specific TRC decay data in the mixing zone and the zone of initial dilution and submit the data to the department for consideration. The following TRC mass balance equation is used to solve for C_d.

$$C_r*Q_s + C_d*Q_d = C_s*(Q_s + Q_d) + Q_s*X_s + Q_e*X_e \tag{7.1-1}$$

Where:

C_r = Background TRC concentration upstream from the outfall, mg/L
Q_s = Q_r*(MZ or ZID) = Stream flow in the regulatory MZ or ZID, cfs
Q_d = Discharge flow, cfs (ADW for concentration limits, AWW for mass limits)
MZ = Stream flow fraction in the regulatory mixing zone
ZID = Stream flow fraction in the zone of initial dilution
C_d = WLA TRC concentration, mg/L

C_s = TRC criteria (acute or chronic criteria, mg/L)
X_s = Chlorine demand of stream water, default value of 0.0 mg/L
X_e = Chlorine demand of discharge flow, default value of 0.0 mg/L

Input parameters:

7Q10 = 2.62 cfs
1Q10 = 1.915 cfs
$C_r = 0.0$ µg/L
Q_r *MZ = 0.25(7Q10) = 0.25(2.62) = 0.655 cfs
Q_r *ZID = 0.025 (1Q10) = 0.025(1.915) = 0.0479 cfs
$Q_d = 0.085$ mgd (0.131 cfs)
$X_s = 0.0$ µg/L
$X_e = 0.0$ µg/L
$C_s = 11$ µg/L chronic criterion
$C_s = 19$ µg/L acute criterion

Step 1: Calculate the chronic wasteload allocation ($WLA_{chronic}$)

C_r*Q_r*MZ + C_d*Q_d = C_s(Q_r*MZ + Q_d)
(0.0)0.655 + C_d(0.131) = 11(0.655 + 0.131)
0 + C_d(0.131) = 11(0.786)
C_d = 8.646/0.131
C_d = 66 µg/L = 0.066 mg/L ($WLA_{chronic}$)

Step 2: Calculate the acute wasteload allocation (WLA_{acute})

C_r*Q_r*ZID + C_d*Q_d = C_s(Q_rZID + Q_d)
(0.0)0.0479 + C_d(0.131) = 19(0.0479 + 0.131)
0 + C_d(0.131) = 19(0.1789)
C_d = 3.3991/0.131
C_d = 25.95 µg/L = 0.02595 mg/L (WLA_{acute})

Step 3: Convert WLAs to Permit Limits Using Permit Derivation Procedure

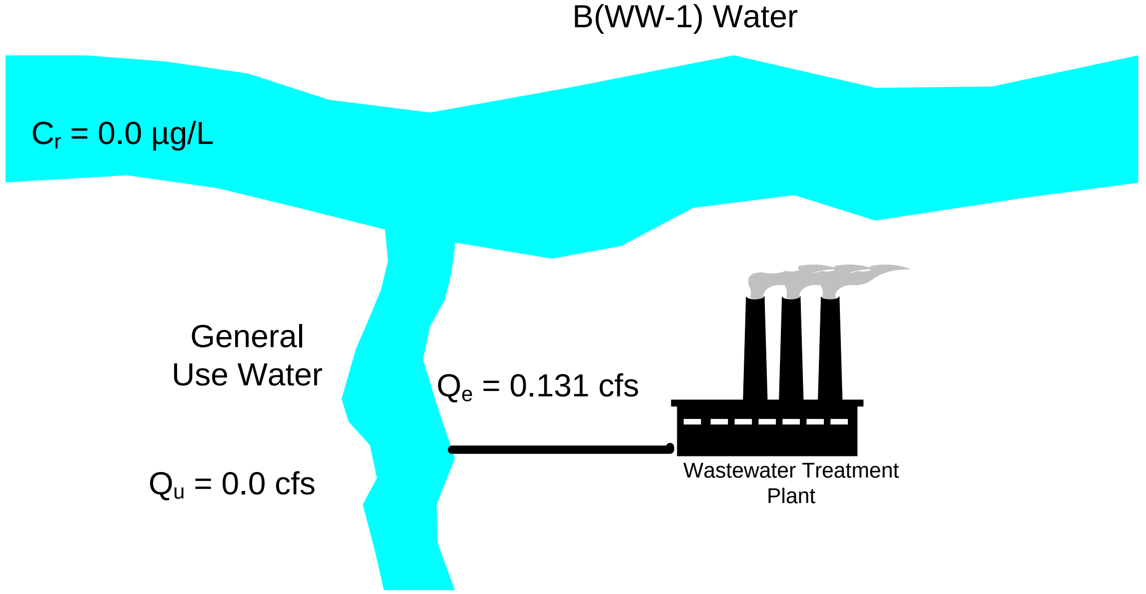
In this example the sampling frequency is once per week. The final monthly and daily maximum limits = 0.02595 mg/L since the acute WLA is governing.

Example 2: Discharge to a General Use First then to Class B(WW-1) Designated Water

KEY:

7Q10 = 7-day 10-year low flow
1Q10 = 1-day 10-year low flow
C_r = Background TRC concentration, µg/L
Q_r*AMZ = Stream flow in the far field ambient MZ of the designated Class B(WW-1) stream, cfs
Q_r*AMZ + Q_d = Total flow in the far field ambient MZ of the downstream designated water, cfs
C_s = TRC acute or chronic criterion, µg/L
Q_u = Background or upstream flow in the general use segment, cfs
Q_e = Discharge flow to the general use segment, cfs
C_e = TRC WLA for the outfall
C_d = TRC WLAs for the Class B(WW-1) stream, µg/L
Q_d = Discharge flow to the Class B(WW-1) water, cfs

$7Q_{10}$ = 2.62 cfs
$1Q_{10}$ = 1.915 cfs



Step 1 and 2 are the same as Example 1.

Step 3: The WLA chronic or acute for ADW flow and WLA chronic or acute for AWW flow from the above step 2 are used in the TRC decay equation. For this example, the WLA_{acute} value of 25.95 µg/L is used to illustrate the procedure. The TRC decay over time "t" is used to calculate the upstream concentration (C_e). The following TRC decay equation for an upstream general waterway is used for solving for C_e.

$$C_e = C_d * e^{(k*t)} \qquad (7.1\text{-}2)$$

Where:
C_e = TRC discharge concentration, μg/L
C_d = TRC WLA for protection of designated water, μg/L
k = Decay rate constant, day $^{-1}$
t = Time of travel in modeled reach, day
L = Length of the general use segment, ft
V = Flow velocity in the general use segment, ft/day

TRC Decay for Upstream General Use Segment

Where:
$C_d = WLA_{acute}$ = 25.95 μg/L
k = 20 day $^{-1}$
t = L/V = 2800/0.2 = 0.162 day

$$\begin{aligned} C_e &= C_d e^{(k*t)} \\ &= 25.95e^{(20)(0.162)} \\ &= 25.95(25.54) \\ C_e &= 663\ \mu g/L = 0.663\ mg/L\ (WLA_{acute}) \end{aligned}$$

Following the above TRC decay calculation, C_e for the chronic wasteload allocation at the outfall ($WLA_{chronic}$) is 1,685 μg/L.

Step 4: Calculate the WLA for the protection of the general use segment.

Based on the ½ 96-hour LC_{50} data available for the most sensitive representative species, the acute WLA for TRC for the protection of the general use segment is 53 μg/L.

Step 5: Select the most stringent WLAs for the protection of downstream uses.

The most stringent acute WLA is 53 μg/L, i.e. the protection of the general use is governing.
The most stringent chronic WLA is 1,685 μg/L.

Step 6: Convert WLAs to Permit Limits Using Permit Derivation Procedure.

In this example the sampling frequency is once per week. The final monthly and daily maximum concentration limits = 53 μg/L due to the acute WLA governing.

8.0 Chloride and Sulfate

The chloride and sulfate criteria are included in 567 IAC 61.3(3). Both the chloride and sulfate criteria are hardness dependent. Chloride criteria also depend on sulfate concentrations and sulfate criteria depend on chloride concentrations. Thus, it is necessary to determine the effluent and ambient background water chemistry parameters before the applicable water quality criteria can be applied.

8.1. Statewide Default Water Chemistry Values

Chloride and sulfate toxicity are both heavily dependent on water hardness. To a lesser degree, chloride toxicity is dependent on the sulfate concentration of the waters, while sulfate toxicity is dependent on the chloride concentration in the water. For those situations where site-specific water chemistry values are not available, statewide default water chemistry values are used. The background values will be updated periodically as new ambient monitoring data become available and will be used to derive the chloride and sulfate WLAs.

8.2. WLAs for Chloride and Sulfate

WLAs for point source discharges in regard to chloride and sulfate are calculated in the same manner as those pollutants listed in 567 IAC 61.3(3) Table 1. The acute WLA is calculated with the use of the 1Q10 stream flow and applied at the boundary of the ZID. The chronic WLA uses the 7Q10 stream flow in its calculation and is applied at the end of the MZ. The sulfate criterion is a single value criterion and is applied at both the end of MZ and the ZID.

A simple mass balance of pollutants is used to meet these boundary conditions.

$$C_r{*}Q_r{*}\ \mathit{MZ\ or\ ZID} + C_e{*}Q_e = C_s{*}(Q_r\ {*}\mathit{MZ\ or\ ZID} + Q_e) \qquad (8.2\text{-}1)$$

where:

C_r = Receiving Waterbody Ambient Background concentration, mg/L

Q_r = Design Stream Low Flow (7Q10 for chronic WLA, 1Q10 for acute WLA)

Q_e = Design Effluent flow, cfs (ADW for concentration limits, AWW for mass limits)

Q_r*(MZ or ZID) = Stream flow in the MZ or ZID, cfs

Q_r*(MZ or ZID)+Q_e = Total flow in the MZ or ZID, cfs

C_e = WLA concentration (or allowed discharge concentration), mg/L

C_s = Applicable water quality standard, mg/L (acute or chronic criteria)

This equation is solved four times for C_e for ADW_{acute}, $ADW_{chronic}$, AWW_{acute}, and $AWW_{chronic}$ WLAs. The results include WLAs for the protection of the acute water quality criteria and WLAs for the protection of the chronic water quality criteria. These WLA values are then carried forward to the Permit Derivation Procedure to derive the water quality based limits for chloride and sulfate.

8.3. The Use of Site-Specific Water Chemistry Data

When site-specific water chemistry data are available, a mass balance equation is used to calculate the water chemistry values in the ZID as shown in Equation 8.3-1:

$$ZID\ water\ chemistry = [(Q_r * ZID * C_r) + (Q_e * C_e)]/(Q_r * ZID + Q_e) \qquad (8.3\text{-}1)$$

Where:

Q_r = Stream 1Q10 Flow, cfs

C_r = Median Background Water Chemistry Concentration, mg/L

Q_e = Discharge Flow, cfs

C_e = Median Discharge Water Chemistry Concentration, mg/L

The ZID water chemistry calculated from Equation 8.3-1 is then used to derive the acute criterion of C_s. For all discharges, the water chemistry values used to calculate the water quality criteria at the boundary of the MZ default to the statewide background values unless site-specific upstream background water chemistry values are provided by the facility. When site-specific water chemistry data are available, the median stream background water chemistry concentration (in mg/L) is used to calculate the water quality criteria at the boundary of the MZ. By inserting the site-specific water quality criteria C_s in Equation 8.2-1, the corresponding WLAs for chloride and sulfate are determined.

9.0 Bacteria

This section describes the methodology to derive wasteload allocations for bacteria for point source discharges that often have a major impact at critical low stream flow conditions. Thus, the Section will only discuss the *E. coli* limits for continuous discharges and controlled discharge lagoons. Other intermittent discharges such as CSOs and MS4s may be rainfall driven and the water quality based limits are developed based on site-specific information on a case by case basis using scientifically defensible methodology.

567 IAC 61.3(3) “a”(1) shows the *E. coli* criteria table that are applicable to Class “A” designated use waters. Waters designated as Class “A1,” “A2,” or “A3” in 567 IAC 61.3(5) are to be protected for primary contact recreational, secondary contact recreational, and children’s recreational uses, respectively.

9.1. Continuous Discharges and Controlled Discharge Lagoons

The *E. coli* criteria are listed in 567 IAC 61.3(3) “a”(1). Table 9.1-1 shows the 30-day geometric mean limits for continuous discharge facilities and controlled discharge lagoon facilities. Note that the final *E. coli* limits for a specific facility may be adjusted due to the consideration of decay as discussed later.

Table 9.1-1. *E. coli 30-day Geomean Limits for Continuous Discharge facilities and Controlled Discharge Lagoon facilities*

Recreational Uses	**Monthly Geometric Mean Limit (org/100 mL)**
Class A1 or A3	126
Class A2	630

9.2. *E. coli* Decay Rate

The *E. coli* decay rate equation is used when there is a discharge to a non-Class "A" stream reach, a storm sewer, a discharge pipe, or tile line which then flows directly into a Class "A" designated use reach. The decay equation projects the amount of *E. coli* loss along the non-Class "A" stream reach, storm sewer, discharge pipe, or tile line. The decay model uses a first-order equation in which the time of travel is incorporated into the calculations. The model formulated in the EPA publication, "Rates, Constants and Kinetics Formulation in Surface Water Quality Modeling" (Second Edition), June 1985 (U.S. EPA [2], 1985), is used for *E. coli* decay. The resulting WLA is the *E. coli* decay plus the WLA calculated for a direct discharge to the designated Class "A" reach. The following *E. coli* equation is used when there is no background flow in the non-Class "A" water, storm sewer, discharge pipe, or tile line, solving for C_d.

$$C_d = C_e e^{(k*t)} \qquad (9.2\text{-}1)$$

Where:

C_d = Allowable *E. coli* discharge concentration, org/100 mL

C_e = Geometric Mean WQS for Class "A" water, 126 org/100 mL for Class "A1" or "A3"; 630 org/100 mL for Class "A2"

k = Decay rate constant, day^{-1}

t = Time of travel in modeled reach, day

EPA 1985 Modeling Study (U.S. EPA [2], 1985) summarizes 12 decay rates for streams and rivers, and six decay rates for lakes and ponds. The decay rates came from studies conducted from the 1920s to the 1980s. The department analyzed the decay rate data published in the EPA 1985 Modeling Study (U.S. EPA [2], 1985) and focused on both stream/river and pond decay rates. The department also reviewed recent studies on bacteria decay rates (Anderson, K.L., et al., 2005; USGS, 2001-2002; U.S. EPA[1], 2001; Sinton, L.W., 2002). The decay rates from all the studies are normalized to a standard temperature of 20°C. The analysis of the combined dataset of the recent studies and the data from EPA 1985 Modeling Study indicates that the median value for bacteria decay rate is 1.0/day. Thus, the bacteria decay rate of 1.0/day at 20 °C is used.

10.0 Thermal Discharges (Temperature WLA)

This section describes the temperature criteria and its implementation procedure. It also presents different alternative options that can be used in lieu of the default approach. Wasteload allocations for temperature are calculated for wastewater discharges resulting from processes that add heat. Biological treatment systems do not add external heat and are sensitive to temperature. Conventional mesophilic bacteria has been shown to perform most optimally when the reactor wastewater temperature is maintained between 78°F and 95°F (26°C and 35°C). Nitrifying bacteria have an even tighter range of optimal reactor temperature 85°F and 92°F (29°C and 33°C) (Cruikshank, C.L. and D. G. Gilles, 2007). In addition, biological treatment systems have relatively constant flow and constant temperature discharges. Thus, temperature criteria are not applicable to biological wastewater treatment facility discharges.

Definitions of stream conditions:

For the purpose of temperature criteria implementation, the following general definitions of different stream conditions apply:

Effluent-Created Streams: The entire flow of the stream consists of effluent flow under normal base flow conditions.

Effluent-Dominated Streams: More than 50% of the stream flow is contributed by effluent flow under normal base flow conditions. These streams usually have a zero 7Q10 flow and are usually considered perennial streams.

Effluent-Supplemented Streams: The effluent flow is less than 50% of the total stream flow under normal base flow conditions. These streams usually have a 7Q10 above zero.

10.1. Temperature Criteria

The temperature water quality criteria are included in 567 IAC 61.3(3) "b"(5) and are summarized below:

1. $\Delta T\uparrow$(allowable temperature rise): Temperature shall not be increased more than 3°C (5.4 °F) for warm water streams and 2°C (3.6°F) for cold water streams.
2. T_{max}: Maximum temperature criteria. For all warm waters except the Mississippi River and cold waters, the allowable maximum temperatures are 32°C and 20°C, respectively. These criteria apply at all times. For the Mississippi River, the allowable maximum

temperature varies by month, and there are two allowable maximum temperature values: one is the absolute maximum temperature never to be exceeded, and the other prohibits exceedance for more than 1% of the hours (86.4 hours) in a 12-month period ending with any month. The 1% of hours allowable maximum temperature is referred to as T_{cap} in this document.

(3) ΔT/hour: Rate of change. The rate of temperature change shall not exceed 1°C/hour.

In addition, 567 IAC 61.2(5) “a” and 567 IAC 61.2(5) “b” include the following statements:

a. The allowable 3°C temperature increase criterion for warm water interior streams, 61.3(3) “b” (5)“1”, is based in part on the need to protect fish from cold shock due to rapid cessation of heat source and resultant return of the receiving stream temperature to natural background temperature. On low flow streams, in winter, during certain conditions of relatively cold background stream temperature and relatively warm ambient air and groundwater temperature, certain wastewater treatment plants with relatively constant flow and constant temperature discharges will cause temperature increases in the receiving stream greater than allowed in 567 IAC 61.3(3) “b”(5) “1.”

b. During the period November 1 to March 31, for the purpose of applying the 3°C temperature increase criterion, the minimum protected receiving stream flow rate below such discharges may be increased to not more than three times the rate of flow of the discharge, where there is reasonable assurance that the discharge is of such constant temperature and flow rate and continuous duration as to not constitute a threat of heat cessation and to not cause the receiving stream temperature to vary more than 3°C per day.

10.2. Heat Transfer Theory

Note: Iowa’s temperature criteria are expressed in Celsius. Effluent temperature limits are often expressed as Fahrenheit. The conversion formula is shown in Equation 10.2-1.

$$ {}^{\circ}\mathrm{F} = {}^{\circ}\mathrm{C} * \frac{9}{5} + 32{}^{\circ}\mathrm{C} \qquad (10.2\text{-}1) $$

In any heat transfer situation, the amount of heat gained or lost may be mathematically defined as:

$$ H = mc\Delta T \qquad (10.2\text{-}2) $$

Where:

H = heat gained or lost (BTU)
m = Mass of body gaining or losing heat (lb)
c = Specific heat (BTU/lb/°F)

ΔT = Temperature change, °F

For simplicity in water quality calculations, the mass (m) of the stream or wastewater is expressed as a flow rate (Q) and is expressed either in terms of million gallons per day (MGD) or cubic feet per second (cfs). The specific heat (c) of water is 1 BTU, or British Thermal Unit, defined as "the amount of heat required to raise one pound of water by 1°F". Incorporating a flow rate instead of mass results in the rate of heat transfer or rate of heat rejection. Equation 10.2-3 incorporating the appropriate conversion factor is:

$$H = nQ\Delta T \qquad (10.2\text{-}3)$$

Where:
Q = Flow rate, cfs or MGD
n = Conversion factor

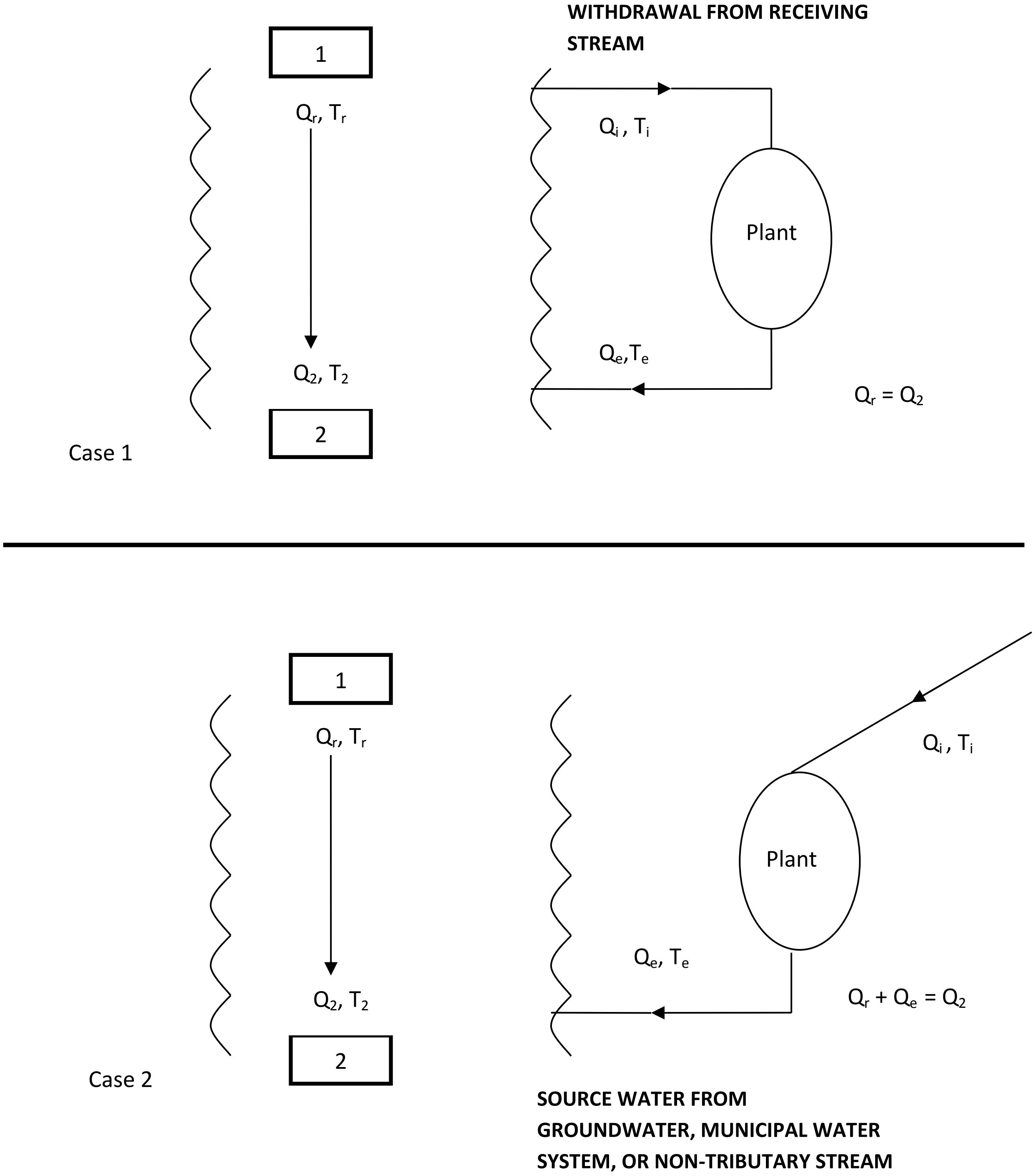
Thus, the amount of heat gained or lost is determined from the heat transfer equation:

$$(Q\Delta T)\text{ gained} = (Q\Delta T)\text{ lost} \qquad (10.2\text{-}4)$$

For the purpose of establishing effluent limits, thermal discharges fall into one of two categories based upon the source of cooling water, as illustrated by Figure 10.2-1. Case 1 situations are those where the source of cooling water is the receiving stream upstream from the point of discharge. Case 2 situations are those where the source of cooling water is not the receiving stream, but instead is a municipal water system, a well, or a different waterbody.

The effluent temperature based limits for Case 1 and Case 2 are calculated based on the heat transfer theory described above.

Figure 10.2-1


WITHDRAWAL FROM RECEIVING STREAM
1
Q_r, T_r
Q_i , T_i
Plant
Q_e,T_e
Q_2, T_2
$Q_r = Q_2$
2
Case 1
1
Q_r, T_r
Q_i , T_i
Plant
Q_e, T_e
$Q_r + Q_e = Q_2$
Q_2, T_2
2
Case 2
SOURCE WATER FROM GROUNDWATER, MUNICIPAL WATER SYSTEM, OR NON-TRIBUTARY STREAM

10.3. Temperature-based Effluent Limits

Temperature-based permit limits are calculated using mass balance equations of the thermal inputs. Formulas for calculating permit limits for the T_{max} and $\Delta T\uparrow$ criteria are as follows:

$$T_{max} \text{ calculation: } (Q_e + Q_r{*}MZ){*}T_{max} = Q_r * MZ * T_r + Q_e{*}\ T_e \tag{10.3-1}$$

$$\Delta T\uparrow \text{ calculation: } T_e = T_r + 3°C * (Q_e + Q_r{*}MZ)/Q_e \tag{10.3-2}$$

Where:

Q_r = Design stream flow 7Q10 in cubic feet per second (cfs) and it must be adjusted for any withdrawal of flow to the facility (intake flow)
T_{max} = Maximum temperature criterion (°C)
T_r = Ambient background temperature (°C)
MZ = Allowable mixing zone fraction, dimensionless, 0-1
$\Delta T\uparrow$ = Allowable temperature increase at the edge of the mixing zone
Q_e = Facility daily maximum discharge flow in million gallons per day (MGD)
T_e = Allowable discharge temperature for the facility (°C)

The allowable temperature increase is for the purpose of maintaining a well-rounded population of warm water fish, and to protect fish that are acclimated to the warmer temperature as a result of the discharge from cold shock due to rapid cessation of heat source from the discharge and resultant return of the receiving stream temperature to natural background temperature. T_{max} is the upper incipient temperature allowable for fish to survive.

The following MZ percentages are used to implement the temperature criteria in WLAs. The MZ used in the temperature WLA calculations varies with the dilution ratio. The facility is separated into one of three categories based on the receiving stream 7Q10 and the design discharge flow:

(1) The MZ is 100% of the 7Q10 flow when the dilution ratio of stream flow (or 7Q10) to discharge flow is less than or equal to 2:1;
(2) The MZ is 50% of the 7Q10 flow when the dilution ratio of stream flow (7Q10) to discharge flow is less than or equal to 5:1 and greater than 2:1;
(3) The MZ is 25% of the 7Q10 flow when the dilution ratio of stream flow (7Q10) to discharge flow is greater than 5:1

Facilities have the option to submit site-specific MZ study data, either from field studies or modeling, such as the use of the CORMIX model, to the department for consideration in lieu of the above default MZ values. MZ study data is collected based on the procedure described in the Mixing Zone Procedures section of this document).

The default maximum mixing zone length for thermal discharges is 2000 feet. If the site-specific thermal plume for a facility can be accurately modeled or measured, the maximum mixing zone length can be longer than 2000 feet as long as the allowable default mixing zone percentages described in this section are met.

Implementation of 567 IAC 61.2(5) "a" and 567 IAC 61.2(5) "b"

According to 567 IAC 61.2(5) "a" and 567 IAC 61.2(5) "b", during the period from November 1 to March 31, for the purpose of applying the 3°C temperature increase criterion, the minimum protected receiving stream flow rate below some discharges may be increased to not more than three times the rate of flow of the discharge, where there is reasonable assurance the discharge is of such constant temperature, flow rate and continuous duration as to not constitute a threat of heat cessation and not cause the receiving stream temperature to vary more than 3°C per day. This is implemented as follows:

(1) If there is a reasonable assurance the discharge is of such constant temperature and flow rate and continuous duration, when the receiving stream flow is less than two times the discharge flow, a stream flow of two times the discharge flow rate is used in lieu of 7Q10.

(2) This procedure applies only when calculating temperature limits for discharges into interior warm water streams and does not apply to discharges to cold water streams or the Mississippi or the Missouri Rivers.

Warm Water Interior Streams and the Big Sioux River

Monthly average, daily maximum, and rate of temperature change effluent limits are calculated based on 567 IAC 61.3(3) "b"(5) "1", 61.2(5) "a" and 61.2(5) "b" temperature criteria.

The 90th percentile monthly background temperatures are used in the calculation of T_{max} limits and the maximum monthly background temperatures are used in the calculation of the $\Delta T\uparrow$ limits. The background temperatures will be updated periodically as new ambient monitoring data become available.

Monthly Average Limits:

WQS state that "no heat shall be added to interior streams or the Big Sioux River that would cause an increase of more than 3°C". This criterion applies at the end of the mixing zone, which is a percentage of the 7Q10 flow (from April to October) or the greater between the 7Q10 and

$2*Q_e$ flow (from November to March), in the receiving stream. The calculation is described by Equation 10.3-3:

$$T_{e\text{-average}} = T_r + 3°C *(Q_e + Q_r*MZ)/Q_e \qquad (10.3\text{-}3)$$

Where:

$T_{e\text{-average}}$ = Allowable Average effluent temperature, (°C)
T_r = Ambient monthly <u>maximum</u> background temperature (°C)

Daily Maximum Limits:

WQS state that "in no case shall heat be added in excess of that amount that would raise the stream temperature above 32 °C". The same MZ and daily maximum discharge flow Q_e is used to calculate monthly average limits and daily maximum limits. The calculation is described by Equation 10.3-4:

$$T_{e\text{-max}} = T_r + (32 °C - T_r) * (Q_e + Q_r*MZ)/Q_e \qquad (10.3\text{-}4)$$

Where:

$T_{e\text{-max}}$ = Maximum allowable effluent temperature, (°C)
T_r = Ambient monthly <u>90th Percentile</u> background temperature (°C)

Rate of Change (ΔT) Limits:

WQS state that "the rate of temperature change shall not exceed 1°C per hour". The criterion is applied to prevent cold shock during plant start up or shut down. A narrative special condition is included in the NPDES permit as stated below to implement the rate of change standard:

"Cessation of thermal inputs to the receiving water by a thermal discharge shall occur gradually so as to avoid fish mortality due to cold shock during the winter months (November through March). The basis for this requirement is to allow fish associated with the discharge-heated mixing zone to acclimate to the decreasing temperature. Likewise, when the discharge resumes the temperature shall be increased gradually to avoid negative impacts to aquatic life in the receiving stream."

Meeting the rate of change requirements by changing receiving water temperature gradually prevents cold shock during the winter season.

Effluent-Created Streams:

The 3°C rise and 1°C per hour change temperature criteria do not apply to effluent-created streams. The 3°C rise and 1°C per hour change are relative to the background ambient temperatures, which cannot be measured for effluent-created streams. The 3°C rise and 1°C per hour change temperature criteria apply to effluent-dominated streams and effluent-supplemented streams.

Cold Water Streams:

The procedures for calculating temperature limits for discharges to cold water streams are the same as those for warm water streams except for the following differences:

(1) Criteria: 567 IAC 567 61.3(3) “b”(5) “2” states that no heat shall be added to streams designated as cold water fisheries that would cause an increase of more than 2°C. In no case shall heat be added in excess of that amount that would raise the stream temperature above 20°C. The 3°C $\Delta T\uparrow$ and 32°C T_{max} criteria for warm water streams are replaced by 2°C and 20°C, as shown in Equations (10.3-5) and (10.3-6).

(2) Stream design low flow, Q_r: 7Q10 is the allowed stream design low flow for Q_r for all seasons.

$$T_{e\text{-}average} = T_r + 2°C * (Q_e + Q_r * MZ)/Q_e \qquad (10.3\text{-}5)$$

Where:

$T_{e\text{-}average}$ = Allowable Average effluent temperature, (°C)

T_r = Ambient monthly maximum background temperature (°C)

$$T_{e\text{-}max} = T_r + (20\ °C - T_r) * (Q_e + Q_r * MZ)/Q_e \qquad (10.3\text{-}6)$$

Where:

$T_{e\text{-}max}$ = Maximum allowable effluent temperature, (°C)

T_r = Ambient monthly 90th Percentile background temperature (°C)

The Missouri River

The procedures for calculating temperature limits for discharges to the Missouri River are the same as for warm water streams except the 7Q10 is the allowable stream design low flow for all seasons.

The Mississippi River

Monthly average, daily maximum, and rate of change limits are calculated according to the temperature criteria described in 567 IAC 61.3(3) “b”(5)“5”. An additional criterion for the Mississippi River is that the water temperature shall not exceed the maximum limits shown in Table 10.3-1 during more than 1 percent of the hours in a 12-month period ending with any month. The 90th percentile background temperatures are used in the calculation of the T_{max} and T_{cap} limits and the maximum monthly background temperatures are used in the calculation of the $\Delta T\uparrow$ limits.

Monthly Average Limits:

WQS state that "no heat shall be added to the Mississippi River that would cause an increase of more than 3°C". This criterion applies at the end of the regulatory MZ. The default MZ is calculated based on the dilution ratio unless site-specific MZ data are available. The calculation is described by Equation 10.3-7:

$$T_{e\text{-}average} = T_r + (3°C) * (Q_e + Q_r\ MZ)/Q_e \qquad (10.3\text{-}7)$$

Where:

T_r = Ambient monthly maximum background temperature (°C)

Daily Maximum Limits:

WQS state that at no time shall the water temperature exceed the maximum limits in Table 10.3-1 by more than 2°C. The same MZ and daily maximum discharge flow Q_e is used to calculate the monthly average limits and daily maximum limits. The calculation is described by Equation 10.3-8:

$$T_{e\text{-}max} = T_r + [2°C + (T_{II} \text{ or } T_{III}) - T_r] *(Q_e + Q_r * MZ)/Q_e \qquad (10.3\text{-}8)$$

Where:

T_{II} = T_{cap} criterion for Zone II (°C)
T_{III} = T_{cap} criterion for Zone III (°C)
T_r = Ambient monthly 90th Percentile background temperature (°C)

Table 10.3-1: *Maximum Allowed River Temperatures Set for Mississippi River Zones II & III (River temperature not to exceed the maximum values in the table below for more than 1 percent of the hours in a 12-month period ending with any month, or by more than 2°C at any time)*

Month	**Zone II**	**Zone III**
	Temperature (°C) = T_{II}	**Temperature (°C) = T_{III}**
Jan.	4	7
Feb.	4	7
Mar.	12	14
Apr.	18	20
May	24	26
Jun.	29	29
Jul.	29	30
Aug.	29	30
Sep.	28	29
Oct.	23	24
Nov.	14	18
Dec.	9	11

Daily Maximum Limits based on T_{cap}*:*

The 1% of hours allowable maximum temperature for the Mississippi River is referred to as T_{cap} in this document. For the Mississippi River, the allowable T_{cap} maximum temperature varies by month. This criterion prohibits T_{cap} to be exceeded for more than 1% of the hours (86.4 hours) in a 12-month period ending with any month.

The temperature limit calculation is described by Equation 10.3-9:

$$T_{e\text{-}max1\%} = T_r + (T_{II\ or\ III} - T_r)*(Q_e + Q_r*MZ)/Q_e \qquad (10.3\text{-}9)$$

Where:

$T_{e\text{-}max1\%}$ = Allowable effluent temperature to meet T_{cap} (°C)
T_r = Ambient 90th Percentile background temperature (°C)

Hourly effluent temperature monitoring data will be used to check compliance with the above temperature limits unless the data are not available.

Rate of Change ($\Delta T\uparrow$) Limits:

WQS state that "the rate of temperature change shall not exceed 1°C per hour". This criterion is implemented as a special condition in the NPDES permit as follows:

"Cessation of thermal inputs to the receiving water by a thermal discharge shall occur gradually so as to avoid fish mortality due to cold shock during the winter months (November through March). The basis for this requirement is to allow fish associated with the discharge-heated mixing zone to acclimate to the decreasing temperature. Likewise, when the discharge resumes the temperature shall be increased gradually to avoid negative impacts to aquatic life in the receiving stream."

The effluent temperature limit based on meeting the $\Delta T\uparrow$ criterion of 3°C (5.4°F) is used as the monthly average limit and the temperature limit based on meeting the T_{max} criterion is used as the daily maximum limit.

Each facility has the option to collect site-specific ambient background temperature data and submit the data to the department for consideration. If site-specific ambient background temperature data are acceptable, they will be used to derive the effluent limits in lieu of default background temperature values.

10.4. Alternative Approaches for the Implementation of Temperature Criteria

The following Section discusses the alternative approaches for the implementation of temperature criteria. The alternative approaches include the use of monthly critical low flows and site-specific data collection. The alternative temperature limits will be applied upon the facility's request.

10.4.1. Monthly or Seasonal Stream Low Flows

Annual critical low flows are applied to calculate temperature wasteload allocation as default option since the annual critical low flows are readily available from USGS publications. Upon a facility request, monthly or seasonal critical low flows will be applied in temperature wasteload allocations when the monthly or seasonal critical low flows can be accurately estimated such as in situations where a USGS gage station is located near an outfall. Thus, the monthly or seasonal critical low flows will be used for calculating temperature limits on a case-by-case basis.

10.4.2. Site-specific Data Collection

Facilities have the option to collect site-specific data, or complete a site-specific thermal study and submit the data to the department for consideration. The following Section provides example site-specific data collection to derive effluent temperature limits.

(1) CORMIX Modeling

Water quality based temperature limits are derived based on default MZ percentages unless site-specific MZ data are available. Facilities have the option to conduct CORMIX modeling study to provide site-specific MZ data. CORMIX is broadly accepted as an easy-to-use yet powerful tool for accurate and reliable point source mixing analysis.

(2) Site-specific Background Temperature

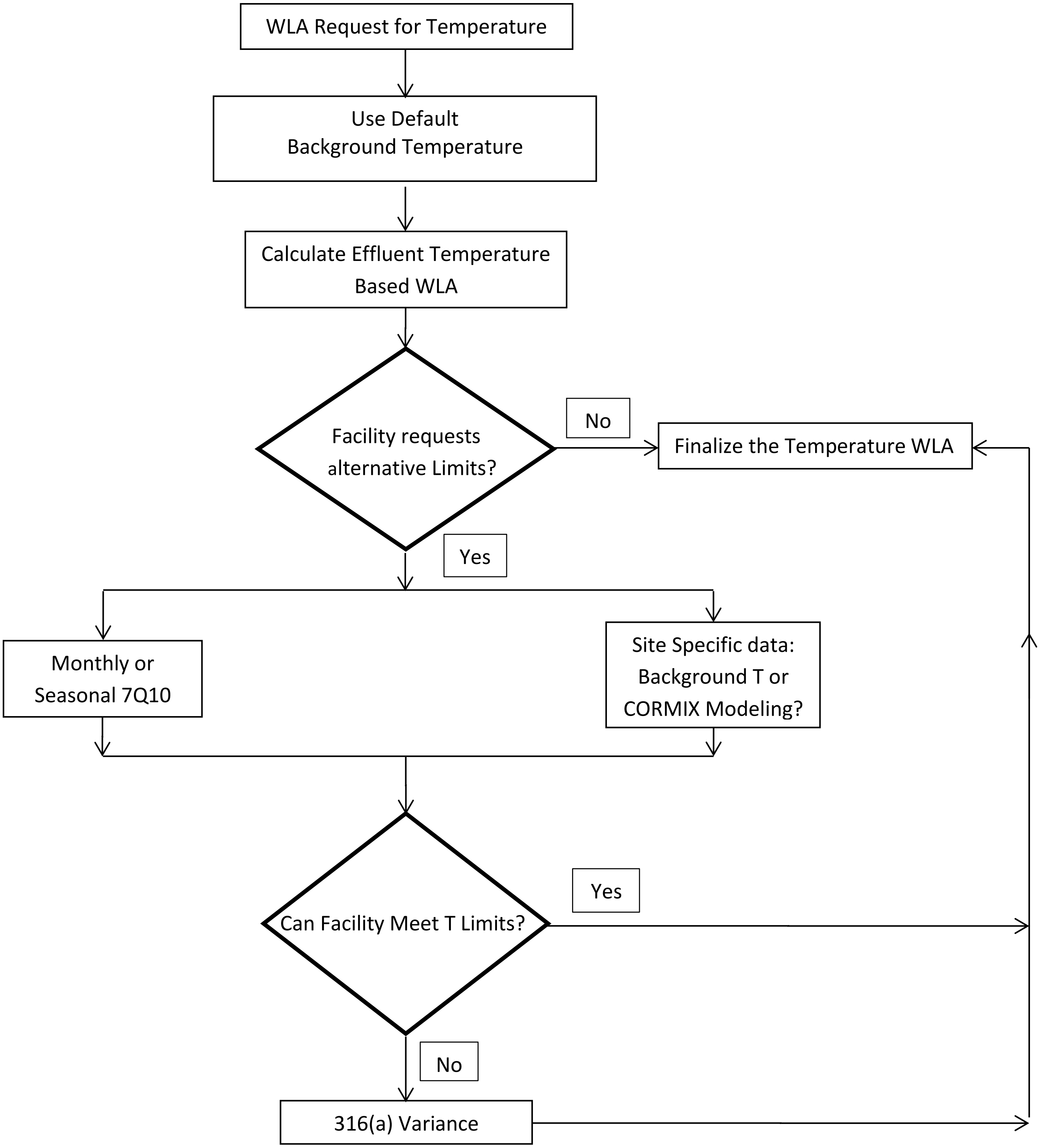
Facilities have the option to collect site-specific upstream background temperature to use in lieu of the default background temperatures in the calculation of temperature limits. In order to establish site-specific background temperature for each month, adequate monitoring data are required. At least 2 years of data for a sampling frequency of once per week is required. The background temperature monitoring data must be representative of the actual background temperature and must not be influenced by the discharge of interest.

10.4.3. 316(a) Demonstrations

Section 316(a) of the Federal Water Pollution Control Act provides that a facility can be granted an alternate thermal effluent limitation if the facility can satisfactorily demonstrate that the effluent limits calculated based on water quality standards are more stringent than necessary to protect a balanced and indigenous community of aquatic organisms in the receiving waterbody. A Section 316(a) demonstration generally requires comprehensive studies which include an evaluation of historical stream and effluent data, characterization of resident species of fish and shellfish populations and predictive impact modeling. A facility with an interest in possible alternate effluent limits based on Section 316(a) must consult the EPA guidance "Interagency 316(a) Technical guidance Manual and Guidance for Thermal Effects Sections of Nuclear Facilities Environmental Impact Statements, 1977" and must contact the department for approval prior to undertaking any studies.

Figure 10.4-1 on the next page shows the temperature criteria implementation process.

Figure 10.4-1 Temperature Criteria Implementation Process Chart



11.0 pH

The pH criterion applies to both Class “A” and Class “B” designated waters, and is described in 567 IAC 61.3(3) “a”(2) and 61.3(3) “b”(2).

In WLAs, pH criteria are calculated to be met at the edge of the MZ. The allowed default MZ dilution for pH is 25% of the 7Q10 flow for interior streams and 10% of the 7Q10 flow for the Mississippi and the Missouri Rivers. Facilities have the option of submitting site-specific MZ data in lieu of the default MZ values through either modeling or a field MZ study as noted in this document.

A pH WLA is calculated based on a mass balance of hydrogen ions by default. The following section explains the methodology.

The equations used to calculate the pH water quality-based limits are shown below:

$$pH\ (WQS) = -\log\ \{(Q_e * 10^{-pHe} + Q_r{*}MZ{*}10^{-pHr})/(Q_e + Q_r{*}MZ)\} \qquad (11.0\text{-}1)$$

where:
Q_e = Design Effluent flow, AWW is used (cfs)
Q_r = Design Stream flow, annual 7Q10 is used (cfs)
pH_e = Allowable Effluent pH, standard unit
pH_r = Ambient receiving waterbody background pH, standard unit
MZ = Mixing zone dilution, dimensionless, between 0-1
pH (WQS) = pH water quality criteria (6.5 to 9.0)

Rearranging Equation 11.0-1:

$$pH_e = -\log\ \{((Q_e + Q_r{*}MZ){*}10^{-pH(WQS)} - Q_r{*}MZ{*}10^{-pHr})/Q_e\} \qquad (11.0\text{-}2)$$

Equation 11.0-2 provides the allowable effluent pH values needed in order to meet the pH criteria in the receiving water at and beyond the edge of the MZ of 6.5 to 9 standard units.

Effluent pH WLAs based on alkalinity and total inorganic carbon are determined by trial and error using the equations described in Appendix A, and will be used if the necessary site-specific data are provided.

12.0 Narrative Water Quality Standards

567 IAC 61.3(2) subsections “a” through “h” specifically mention general water quality criteria that apply to all surface waters including general use and designated use waters. To ensure that narrative criteria for toxicants are attained, the federal Water Quality Standards Regulation

requires States to develop implementation procedures (see 40 CFR 131.11(a)(2)). The criterion at 567 IAC 61.3(2) "d", which states that waters must be free from any substance which is acutely toxic to human, animal or plant life; and the criterion at 567 IAC 61.3(2) "g", which provides for the protection of livestock watering, are discussed in the following subsections.

12.1. General Use Segments (567 IAC 61.3(2) "d")

567 IAC 61.3(1) "a" defines General Use Segments as *"intermittent watercourses and those watercourses which typically flow only for short periods of time following precipitation and whose channels are normally above the water table. These waters do not support a viable aquatic community during low flow and do not maintain pooled conditions during periods of no flow. The general use segments are to be protected for livestock and wildlife watering, aquatic life, noncontact recreation, crop irrigation, and industrial, agricultural, domestic, and other incidental water withdrawal uses."*

In order to ensure acutely toxic conditions are not caused in the stream for a specific chemical of concern a no-effect level based on chemical-specific toxicity is established. The derivation of the no-effect level for a chemical-specific translator is different than the development of a Lethal Concentration (1% mortality) value, LC_1, based on whole effluent toxicity testing data. The development of an LC_1 incorporates the synergistic effects of all chemicals in the effluent. The commonly used method to derive a no-effect concentration for a specific chemical based on the acute toxicity end point of LC_{50} (or Effective Concentration, EC_{50}) is the use of ½ the 96-hour or 48-hour LC_{50} (or EC_{50}) values (U.S. EPA [1], 1985). A No Observed Effect Concentration (NOEC) is also used. The no-effect level for a specific chemical is determined by calculating the value of ½ the 96-hour or 48-hour LC_{50} (or EC_{50}) or a NOEC for the most sensitive representative resident species for the waterbody of interest. The ½ the 96-hour or 48-hour LC_{50} (or EC_{50}) value or NOEC for the most sensitive representative resident species are obtained for the species from the EPA 304(a) criteria document, ECOTOX database or other data sources meeting the credible data requirement in Iowa Code 455B.194. The toxicity data is reviewed by department staff before it is used for the development of WLAs. The no-effect level for the most sensitive representative resident species and associated concentration is used as the narrative criterion translator in the following mass balance equation:

$$Cr*Qr + Ce*Qe = NOEC \text{ or } (\tfrac{1}{2} LC_{50} \text{ or } \tfrac{1}{2} EC_{50})*(Q_r + Q_e) \qquad (12.1\text{-}1)$$

Where:
C_r = Receiving Waterbody Ambient Background concentration, mg/L
Q_r = Design stream low flow in the general use segment (above the outfall), (1Q10 = 0.0 cfs)
Q_e = Design Effluent flow, cfs
LC_{50} = Lethal concentration of a toxicant that would result in 50% mortality of the test organisms in 48 or 96 hours for the most sensitive representative resident species in the general use segment, µg/L
EC_{50} = Effective concentration of a toxicant that causes an observable adverse effect (such as death, immobilization, or serious incapacitation) in 50% of the test organisms in 48 or 96 hours, for the most sensitive representative resident species in the general use segment, µg/L
NOEC = The highest tested concentration of a toxicant at which no adverse effects are observed on the aquatic test organisms in 48 or 96 hours
MZ = Mixing zone dilution, dimensionless, 0-1
C_e = WLA concentration for the pollutant of concern, µg/L

Equation 12.1-1 is solved for C_e. This value is compared to the acute WLA for the same pollutant calculated to protect downstream designated waters described in the previous sections including ammonia nitrogen, total residual chlorine, and all other priority pollutants. The most stringent of the WLAs is carried forward to the Permit Derivation Procedure Section. It is important to note that narrative criteria translator value is applied at the end-of-pipe for General Use waters unless site-specific data prove otherwise. This approach is based on the definition of General Use segments in 567 IAC Chapter 61.3(1)“a”.

12.2. Designated Waters (567 IAC 567 61.3(2) “d”)

For chemicals with numerical water quality criteria in designated use waters the numerical criteria apply at the end of regulatory MZ or ZID in order to prevent acutely toxic conditions. For chemicals without numerical water quality criteria the no-effect level concentration for the most sensitive representative resident species is applied at the end of the ZID as shown in the mass balance shown in Equation 12.2-1:

$$C_r*Q_r *ZID + C_e*Q_e = NOEC \text{ or } (1/2\ LC_{50} \text{ or } ½\ EC50)*(Q_r *ZID + Q_e) \qquad (12.2\text{-}1)$$

Where:

C_r = Receiving Waterbody Ambient Background concentration, mg/L
Q_r = Design stream low flow in the designated use segment (above the outfall), cfs (1Q10)
Q_e = Design Effluent flows, cfs
LC_{50} = Lethal concentration of a toxicant that would result in 50% mortality of the test organisms in 48 or 96 hours for the most sensitive representative resident species in the general use segment, µg/L
EC_{50} = Effective concentration of a toxicant that causes an observable adverse effect (such as death, immobilization, or serious incapacitation) in 50% of the test organisms in 48 or 96 hours, for the most sensitive resident species in the general use segment, µg/L
NOEC = The highest tested concentration of a toxicant at which no adverse effects are observed on the aquatic test organisms in 48 or 96 hours
C_e = WLA concentration for the pollutant of concern, µg/L
ZID = Zone of Initial Dilution, dimensionless, 0-1

Equation 12.2-1 is solved for C_e. This value is compared to the acute WLA for the same pollutant calculated to protect downstream designated waters described in the previous sections including ammonia nitrogen, total residual chlorine, and all other priority pollutants. The most stringent of the WLAs is used in the Permit Derivation Procedure Section.

12.3. Livestock Watering 567 IAC 61.3(2) "g"

For livestock watering, the following cation and anion guideline values are applicable. The guideline values are required to be met at the boundary of the MZ for designated use segments and at the end of pipe for general use segments. Thus, the WLAs for these cations and anions are calculated to meet the guideline values at the end of allowable MZ, which is by default 25% of the 7Q10 stream flow.

Table 12.3-1. *Recommended Water Quality Guidelines for Protecting Defined Uses*

Ions	Recommended Guidelines Values* (mg/L)
Calcium	1,000
Chloride	1,500
Magnesium	800
Sodium	800
Sulfate	2,000
Nitrate+Nitrite-N	100

*: Based on the guidelines for livestock watering.

13.0 Mixing Zone Procedures

As stated in the *EPA Water Quality Standards Handbook* (U.S. EPA[2], 1994), "It is not always necessary to meet all water quality criteria within the discharge pipe to protect the integrity of the water body as a whole." Sometimes it is appropriate to allow concentrations above the water quality criteria in small areas near outfalls. These areas are called mixing zones (MZs). Whether to establish a MZ policy is a matter of State discretion, but any State policy allowing for MZs must be consistent with the federal Clean Water Act. According to EPA's Technical Support Document for Water Quality-based Toxics Control (TSD) (U.S. EPA, 1991*), "a mixing zone is an area where an effluent discharge undergoes initial dilution and is extended to cover the secondary mixing in the ambient waterbody. A mixing zone is an allocated impact zone where water quality criteria can be exceeded as long as acutely toxic conditions are prevented."*

567 IAC 60.2 defines Mixing Zone and Zone of Initial Dilution. *"Mixing zone means a delineated portion of a stream or river in which wastewater discharges will be allowed to combine and disperse into the water body. The chronic criteria of 567¬subrule 61.3(3) will apply at the boundary of this zone."*

"Zone of initial dilution means a delineated portion of a mixing zone in which wastewater discharges will be allowed to rapidly combine and begin dispersing into the water body. The acute criteria of 567¬subrule 61.3(3) will apply at the boundary of this zone."

To ensure MZs do not impair the integrity of a waterbody it is critical to demonstrate that an MZ does not cause lethality to passing organisms, likely pathways of exposure are considered, and that there are no significant human health risks. When wastewater is discharged into a waterbody it goes through two stages, each with distinctive mixing characteristics:

(1) The first stage includes mixing and dilution that are determined by the initial momentum and buoyancy of the discharge. This initial contact with the receiving water is where the concentration of the effluent is at its greatest in the water column. The design of the discharge outfall pipe must provide ample momentum to dilute the concentration in the immediate contact area as quickly as possible.

(2) The second stage of mixing covers a more extensive area in which the effluent's initial momentum and buoyancy is diminished and the discharge is mixed primarily by ambient turbulence.

The objective of the MZ procedure is to provide the methodology used to incorporate the allowable MZ which is used in determining the applicable water quality based effluent limitations for a wastewater discharge.

567 IAC 61.2(4), *Regulatory mixing zones,* states that *"mixing zones are recognized as being necessary for the initial assimilation of point source discharges which have received the required degree of treatment or control. Mixing zones shall not be used for, or considered as, a substitute for minimum treatment technology required by Subrule 61.2(3). The objective of establishing mixing zones is to provide a means of control over the placement and emission of point source discharges so as to minimize environmental impacts."*

The standards contain specific criteria and considerations, which are used in determining the extent and nature of a MZ. The most restrictive of the provisions establishes the allowable MZ dimensions and flow.

The chronic criteria for toxics and ammonia nitrogen are to be met at the boundary of the MZ. The acute criteria for toxics and ammonia nitrogen are to be met at the boundary of the ZID. Although not specifically discussed in the standards, the most critical effects of Biochemical Oxygen Demand (BOD) are not expected to be observed until after the end of the regulatory MZ. This is because the movement of water through the MZ normally occurs faster than the biological uptake of oxygen used in the degradation of BOD. As a result, the lowest dissolved oxygen point due to the point source discharge occurs beyond the MZ boundary. Per 567 IAC 61.2(4) "c", the stream flow used in determining WLAs ensures compliance with the maximum contaminant level as well as the chronic and human health criteria of Table 1 in 567 IAC Chapter 61. These criteria must be met at the boundary of the allowable MZ.

Due to extreme variations in wastewater and receiving water characteristics, spatial dimensions of MZs are defined on a site-specific basis according to 567 IAC 61.2(4) “a”. 567 IAC 567 61.2(4) “d”(4) also allows facilities to collect site-specific MZ data to submit to the department for consideration. A facility that discharges to interior streams, rivers, and the Big Sioux, Mississippi or Missouri Rivers have the option to provide in-stream data that supports a different percentage of the design stream flow contained in the MZ. Any increase in the allowable MZ flow must be consistent with the MZ length restrictions.

The site-specific MZ and ZID are determined in one of two ways, by:

(1) Actual field measurements such as a tracer or dye study at or close to design low stream flow conditions; or by
(2) Use of a dispersion model such as the CORMIX model.

A field procedure protocol has been developed for facilities to obtain actual field data, which is presented in the Mixing Zone Study Field Procedure Protocol Section. Unless site-specific MZ data are submitted for consideration, default regulatory MZs as defined in 567 IAC Chapter 61 are used.

13.1. Mixing Zone Study Field Procedure Protocol (for shoreline discharges)

567 IAC 61.2(4) “d”(4) and 61.2(4) “g”(4) state that a facility may provide instream data to the Department for consideration. The instream data presents the site-specific mixing zone size or the percent of stream critical low flow that mixes with the effluent flow and they are collected through a mixing zone study. The objective of the mixing zone study is to demonstrate the amount of mixing taking place between the effluent flow and the stream flow within the allowed mixing zone (MZ) and zone of initial dilution (ZID) lengths at critical low flow conditions. Both the MZ and ZID are to be measured in a mixing zone study. If only one of the MZ and the ZID is measured, the default percentage of design low flow defined in 567 IAC Chapter 61 is used in the WLA calculations for whichever of the MZ or the ZID that is not measured.

Facilities working on a mixing zone study should submit a mixing zone study plan to the Department for approval, which ensures that all data collection and analysis will be applicable to the WLA calculations.

Mixing zone study data may be collected through a field study and/or plume modeling. If the facility chooses to perform a field study using dye, then the facility is *required* to obtain additional authorization for the use of biodegradable and nontoxic dye by the Department if the use of the dye has the potential to reach any Waters of the State and the use of the dye has not been authorized in the facility's current NPDES permit. The dye's aquatic toxicity and human health effects are to be provided when the request for the use of dye is submitted to the Department. The corresponding department field office must be informed of the mixing zone study by the facility at least 48 hours before the study is conducted. In lieu of the addition of a dye to the effluent, physical or chemical parameters already present in the effluent, such as chloride, temperature, or specific conductivity may be used as a tracer.

The following sections describe the methodologies that are commonly used in a mixing zone study: (1) field mixing zone study, and (2) plume modeling. Other scientifically defensible methodologies may be used in a mixing zone study upon approval by the Department.

A. Field Mixing Zone Study

Preliminary Data Gathering and Setup:

The following items for a distance of 2,000 feet downstream from the outfall of the facility (unless other distance limitation is known to apply) and within 200 feet upstream of the outfall (or other distance that is representative of the upstream background conditions) must be recorded. The information may be recorded using a combination of narrative descriptions, maps, and photos.

(1) Describe the stream substrate (sand, silt, fine or coarse gravel, or rock);
(2) Note dams, diversions, pools, riffles, eddies, snags, obstructions, sand bars, or other features that may affect mixing;
(3) Note the following factors that could reduce the maximum mixing zone length: Public water supply water intakes, other nearby discharges, perennial streams downstream of the outfall, any established recreational areas such as public beaches, and state, county and local parks;
(4) Describe the amount and nature of stream channel meandering;
(5) Describe the outfall and discharge flow at the time of the study such as the size and configuration of splash pools, outfall height or depth, outfall diameter, average velocity of flow exiting outfall, discharge flow in the discharge pipe is under pressure or by gravity, discharge flowing full or partially full, discharge pipe is submerged or not submerged;

Field Study Protocol:
Stream surveys to gather mixing zone data must be collected at low stream flow conditions that are as near to the stream design low flow as is normally feasible during the year. The stream flow must not exceed 20 times the 7Q10 flow unless approved by the Department.

Prior to beginning the field study, the following steps must be followed:

(1) Lay out downstream station locations along the shoreline at intervals of 50 feet, 100 feet, 200 feet, 500 feet, 1,000 feet, 1,500 feet, and 2,000 feet below the outfall. More station locations may be used as necessary. Assemble boundary marking floats or stakes. Test stream depth for float line length and ability to wade;
(2) Measure or record both the receiving stream flow and the discharge flow at the time of the study. The method used in stream flow measurement must be consistent with USGS methodology;
(3) A test run using dye may be used to determine whether 100% of the stream flow is mixed with the discharge flow within the maximum allowed mixing zone length specified in 567 IAC 61.2(4) “b”(2) & “e”(2).

Facilities have the option to consider two different field efforts in obtaining the mixing zone information: Visible Assessment and Tracer Assessment. These are discussed in greater detail below.

a. *Visible Assessment*:
   This procedure is a visible field documentation of the effluent’s mixing with the stream flow under low stream flow conditions. Typically, a facility would add dye to the effluent to facilitate the visible assessment.

The following is a brief summary of the procedures that must be followed for a visible assessment:

(1) Introduce the dye to the effluent at the determined volume/rate and location;
(2) The dye must be used according to the manufacturer’s instructions;
(3) Map the plume width boundaries in the receiving stream by setting out markers at each station location (this process must be carefully planned without interfering with the natural plume);
(4) Determine the MZ length, which is the minimum of the following:
    a. 2,000 feet downstream from the outfall of the facility
    b. Distance to limitation specified in 567 IAC 61.2(4) “b”(2) & “e”(2)

c. Location at which complete mixing is achieved

(5) Measure the stream flow in the dye plume (depth, velocity, cross section) at the boundary of the MZ by dividing the cross section into equidistant intervals;

(6) Measure the total stream flow at the boundary of the MZ (optional if the stream flow above the outfall is measured) by dividing the cross section into at least 10 equidistant intervals (or the number of equidistant intervals specified in the USGS methodology);

(7) Determine the ZID length, which is 10% of the determined MZ length;

(8) Measure stream flow in the dye plume (depth, velocity, cross section) at the boundary of the ZID;

(9) Calculate mixing zone and zone of initial dilution percentages.

$$MZ\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ at\ MZ\ boundary - Discharge\ flow}{Total\ \ stream\ flow\ upstream\ of\ discharge}$$

Or

$$MZ\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ at\ MZ\ boundary - Discharge\ flow}{Total\ Stream\ flow\ at\ MZ\ boundary - \ Effluent\ discharge\ flow}$$

$$ZID\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ at\ ZID\ boundary - Discharge\ Flow}{Total\ stream\ flow\ upstream\ of\ discharge}$$

Or

$$ZID\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ at\ ZID\ boundary - Discharge\ Flow}{Total\ stream\ flow\ at\ ZID\ boundary - \ Effluent\ discharge\ flow}$$

(10) Prepare a report of the findings. Reports must be submitted on the appropriate form supplied by the department.

Specific stream conditions may warrant a modified MZ study plan. Models are available to project the percentages of mixing obtained during field flow conditions to design low flow regime.

b. *Tracer Assessment*:

The objective of this procedure is to provide even greater accuracy in characterizing the mixing of an effluent with the receiving stream by measuring concentrations of a tracer (dye, specific conductivity, chloride, temperature, etc.) at various locations in the MZ. It is preferable to use a tracer that can be measured in the field so that no laboratory analytical work is needed (if laboratory testing is used, more data collection is required than what is described below in order to obtain a detailed map of the plume boundary). The tracer must be measurable to a high level of precision and should be present in the receiving stream and discharge in different amounts. Generally, the difference between the concentration in the effluent plume and the receiving stream should be detectable when the effluent plume has been 90% diluted by stream water. The collection of samples (or measurement of concentrations) must be made across the stream from the shoreline until a point in the stream is reached where the effluent is not expected to be present.

The following procedures must be followed where <u>field measurement of a tracer</u> is used:

(1) Measure the concentration of the tracer in the receiving stream at one or more locations upstream from the outfall that are representative of the background conditions;
(2) Measure the concentration of the tracer in one or more samples of effluent;
(3) Measure the concentration across the stream cross section at a minimum of 20 equidistant intervals and map the plume boundaries in the receiving stream by setting out markers at each station location
    a. If the range of concentrations across the entire channel at a station is less than 10% of the mean it is assumed that complete mixing has occurred
    b. If complete mixing is found at a station, data must be gathered on one or more cross sections between the last two stations in the study to find the location where the effluent is first completely mixed with the flow in the stream
(4) Determine the MZ length, which is the minimum of the following:
    a. 2,000 feet downstream from the outfall of the facility
    b. Distance to limitation specified in 567 IAC 61.2(4) “b”(2) & “e”(2)
    c. Location at which complete mixing is achieved
(5) Measure the stream flow in the tracer plume (depth, velocity, cross section) at the boundary of the MZ by dividing the cross section into equidistant intervals;
(6) Measure the total stream flow at the boundary of the MZ (optional if the stream flow above the outfall is measured) by dividing the cross section into at least 10 equidistant intervals (or the number of equidistant intervals specified in the USGS methodology);
(7) Determine the ZID length, which is 10% of the determined MZ length;
(8) Determine the ZID width boundary by measuring the concentration across the stream at the downstream length boundary of the ZID;

(9) Measure stream flow in the tracer plume (depth, velocity, cross section) at the boundary of the ZID;

(10) Calculate mixing zone and zone of initial dilution percentages

$$MZ\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ at\ MZ\ boundary - Discharge\ flow}{Total\ stream\ flow\ upstream\ of\ discharge}$$

Or

$$MZ\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ at\ MZ\ boundary - Discharge\ flow}{Total\ Stream\ flow\ at\ MZ\ boundary - \ Effluent\ discharge\ flow}$$

$$ZID\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ at\ ZID\ boundary - Discharge\ Flow}{Total\ stream\ flow\ upstream\ of\ discharge}$$

Or

$$ZID\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ at\ ZID\ boundary - Discharge\ Flow}{Total\ stream\ flow\ at\ ZID\ boundary - \ Effluent\ discharge\ flow}$$

(11) Prepare a report of the findings. Reports must be submitted on the appropriate form supplied by the department.

Specific stream conditions may warrant a modified MZ study plan. Models are available to project the percentages of mixing obtained during field flow conditions to design low flow regime.

B. Plume modeling

Please refer to the EPA's Technical Support Document for Water Quality-based Toxics Control (TSD) (U.S. EPA, 1991). Site-specific discharge and stream data must be used in the plume modeling to ensure that the results are representative of the discharge conditions of interest.

C. Use of MZ Study Results:

The department uses the mixing zone study results to calculate water quality based effluent limits. It is required that a mixing zone study be performed by the facility and submitted to the department’s Wasteload Allocation staff prior to NPDES Permit re-issuance. The department has the ability to provide the facility with preliminary water quality-based permit limits to aid in evaluating the need for a mixing zone study. Once the facility submits the mixing zone study results to the department, Wasteload Allocation staff will review mixing zone study results to determine if the data and analysis are applicable to the WLA calculations. If the data and analysis are not usable, then default values described in 567 IAC 61.2(4) for the WLA are used.

13.2. Installation of a Diffuser

An in-stream diffuser to disperse effluent across a more significant portion of the stream is an optional artificial means to achieve rapid mixing. The diffuser shall be properly designed to achieve rapid mixing with the stream flow at the diffuser to minimize the potential adverse impact from the discharge in order to prevent acutely toxic conditions and allow safe fish passage. The wasteload allocation calculations will be based on the percentage of stream critical low flow that passes across the operational portion of the outfall diffuser as demonstrated by a dye study or an accurate modeling of the diffuser. The discharge flow from diffuser ports must achieve homogeneous mixing within 100 feet downstream of the diffuser. Thus, for a dye study or modeling of a diffuser, a rapid and uniform mixing dilution factor has to be achieved within 100 feet downstream of the diffuser. Several permits are required for the diffuser structure, including a construction permit. More detailed information on diffuser design can be found in EPA's Technical Support Document for Water Quality-based Toxics Control (TSD) (U.S. EPA, 1991). Further requirements regarding the use of an effluent diffuser device are found in 567 IAC 61.2(4) “d”(3).

A. Diffuser Study for NPDES Permit Renewal

A diffuser is designed and built to maximize near-field mixing under design low flow conditions. Typically, a diffuser is centered in the low flow channel of the receiving stream. However, as conditions change over time a diffuser may no longer operate as designed. A diffuser study is performed in order to ensure the diffuser is operating as designed. If a diffuser is not operating as designed, maintenance and repairs must be undertaken to return it to design conditions. The following procedures must be followed for a diffuser study. The stream flow must not exceed 20 times the 7Q10 flow unless approved by the Department. There are two options for completing a diffuser study: a visible assessment and plume modeling. Other scientifically defensible methods may also be used.

a. *Visible Assessment:* A visible assessment is a dye study to characterize the operation and mixing achieved by the diffuser in the receiving stream.

    (1) Describe the diffuser at the time of the study such as the elevation of the ports, diameter, discharge velocity, configuration, total number of ports, and number of active ports;
    (2) Measure and record the discharge flow at the time of the study;

(3) Determine the stream bathymetry at the location of the diffuser by measuring the stream depth at a minimum of 20 equidistant intervals across the entire width of the receiving stream;

(4) Lay out a downstream station location along the shoreline at 100 feet below the diffuser. Assemble boundary marking floats or stakes. Test stream depth for float line length and ability to wade;

(5) Introduce the dye to the effluent before the diffuser at the determined volume/rate;

(6) The dye must be used according to the manufacturer's instructions;

(7) Map the plume boundaries in the receiving stream by setting out markers at the station location (100 feet below the diffuser);

(8) Measure the stream flow in the dye plume (depth, velocity, cross section) 100 feet below the diffuser;

(9) Measure either the upstream receiving stream flow or the total stream flow at 100 feet below the diffuser at the time of the study by dividing the cross section into at least 10 equidistant intervals (or the number of equidistant intervals specified in the USGS methodology);

(10) Calculate the percentage of stream flow available for dilution for wasteload allocation calculations

$$Dilution\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ 100\ ft\ downstream\ from\ diffuser - Dischrge\ flow}{Total\ stream\ flow\ upstream\ of\ discharge}$$

Or

$$Dilution\ \% = 100 * \frac{Flow\ in\ the\ dye\ plume\ 100\ ft\ downstream\ from\ diffuser - Dischrge\ flow}{Total\ Stream\ flow\ at\ 100\ ft\ downstream -\ Effluent\ discharge\ flow}$$

(11) Prepare a report of the findings. Reports must be submitted on the appropriate form supplied by the Department.

Specific stream conditions may warrant a modified diffuser study plan. Models are available to project the percentages of mixing obtained during field flow conditions to design low flow regime.

b. *Modeling Assessment:*

Please refer to the EPA's Technical Support Document for Water Quality-based Toxics Control (TSD) (U.S. EPA, 1991). Site-specific discharge and stream data must be used in the plume modeling to ensure that the results are representative of the discharge conditions of interest.

B. Use of Diffuser Study results

The department uses the diffuser study results to calculate water quality based effluent limits. A diffuser study shall be performed by the facility and submitted to the department's Wasteload Allocation staff prior to NPDES Permit re-issuance. Once the facility submits the diffuser study results to the department, Wasteload Allocation staff will review diffuser study results to determine if the data and analysis are applicable to the WLA calculations. If the data and analysis are not usable or a diffuser study is not submitted, then default values described in 567 IAC 61.2(4) for the WLA are used.

C. Annual Diffuser Performance Analysis

The NPDES permittee is required to submit a Diffuser Performance Analysis report to the Department annually. The purpose of the annual Diffuser Performance Analysis is to identify if the diffuser is performing as designed and may not be used for NPDES permitting purposes.

The permittee may choose to use the same procedure described in Part A - *Diffuser Study for NPDES Permit Renewal* or select a simplified approach as described in the Diffuser Special Monitoring Requirements Section of the NPDES permit.

The annual diffuser analysis must be performed at a stream flow as close as possible to stream critical low flow conditions. The stream flow must not exceed 20 times the 7Q10 flow unless approved by the Department. The annual diffuser performance analysis must identify whether all diffuser ports, that were active when the mixing zone percentage used in the current NPDES permit was established, are functioning properly. The annual diffuser performance analysis must also assess if rapid and uniform mixing is occurring within 100 feet downstream of the active diffuser ports with the stream flow as close as possible to critical low flow conditions.

13.3. Special Limitations of Mixing Zones

The following conditions define where the use of the default MZ is <u>not</u> appropriate:

(1) Where drinking water contaminants are of concern, MZs shall not encroach on drinking water intakes (which is consistent with 567 IAC 61.2(4) "b"(2) "2" and "e"(2) "2");

(2) MZs or ZIDs are <u>not</u> allowed for designated lakes and wetlands (which is consistent with 567 IAC 61.2(4) "d"(1) and 61.2(4) "g"(1));

(3) A ZID is <u>not</u> allowed in waters designated as Class B(CW), Cold Water (which is consistent with 567 IAC 61.2(4) "d"(2) and 61.2(4) "g"(2));

(4) For backwater conditions such as those found on the Mississippi River and reservoirs the default MZ and ZID is 0.1% of the 7Q10 and 0.01% of the 1Q10 of the stream flow for toxics and 0.1% of the 30Q10 and 0.01% of the 1Q10 of the stream flow for ammonia nitrogen unless site-specific MZ data are available. Facilities have the option of providing site-specific MZ data through modeling or field study to be used in lieu of the default MZ

(5) For a discharge to a side-channel where the mixing zone and zone of initial dilution are confined to the side-channel, the corresponding critical low flows in the side-channel must be used to derive wasteload allocations in lieu of the total stream critical low flows.

Other mixing zone limitations are established at 567 IAC 61.2(4) “b”(2) and “e”(2).

13.4. Multiple Discharges into the Same Reach

Multiple facilities that discharge into the same reach in close proximity will have overlapping mixing zones for the common pollutant(s) of concern. The development of wasteload allocations for common pollutants is based on the impact to water quality resulting from the combined discharges, the assimilative capacity for the river reach, and the allocation of the assimilative capacity to each facility. Site-specific evaluations are allowed to confirm the extent to which mixing between the two discharges occurs.

Different mechanisms are used to allocate the assimilative capacity for the river reach. For example, (1) to allocate the wasteloads based on the proportion of the existing load contributed by each facility; (2) to allocate the same water quality based concentration limits and proportionately on the mass loading limits based on the average wet weather flow.

If a new or expanded discharge from a facility is proposed in close proximity to an existing facility with overlapping mixing zones, the discharge limits for the existing facility will not be revised significantly to accommodate the new or expanded discharge. A new or expanded facility and existing facility can negotiate conditions that provide for less stringent limits for the new or expanded facility and more stringent limits for the existing facility as long as the water quality standards resulting from the combined discharges are met.

13.5. Downstream Protection

As previously stated, WLAs are calculated to protect all downstream uses. WLAs are performed at the outfall and at the boundary in the route of flow downstream where the designation changes with more stringent criteria. The most stringent WLA between the WLA at the outfall and the WLA at the downstream location governs. Two separate types of situations can occur relating to downstream protection:

Case 1: Designated Use Change on the Same Stream

At the boundary of the designated use change downstream of an outfall, the effluent will be completely mixed with the stream flow. Thus, the WLA is calculated based on one hundred percent mixing with the applicable design stream low flows.

Case 2: Designated Use Change at the Confluence of Two Streams

When one stream joins another stream, ambient mixing occurs. The area of ambient mixing is not a regulatory mixing zone, although the mixing is accounted for in a WLA at the confluence of two streams. Like a regulatory mixing zone, any consideration must ensure that an area of ambient mixing is protective of the designated use(s) of the downstream waterbody as a whole. Applying the regulatory mixing zone rules and procedures to an area of ambient mixing at the confluence of two streams meets this requirement.

14.0 Flow Variable Limitations Procedures

The purpose of flow variable limits is to provide point source discharge facilities the option to have water quality based effluent limits that vary with stream flow. However, flow variable limits are not a replacement for treatment. The approach provides a mechanism that is both protective of water quality standards and provides flexibility for facilities to achieve compliance under elevated stream flow conditions.

In order to be eligible for flow variable limits, a facility is required to:

1. Demonstrate its ability to meet its effluent limits at critical low flow conditions;
2. Be able to obtain daily stream flow measurements that accurately represent the stream flow at the outfall location;
3. Pass an antidegradation review if the facility does not currently have flow variable limits.

Flow variable limits are expressed as pounds per day per unit stream flow in cubic feet per second (lbs/day/cfs) and are based on the applicable chronic and acute water quality criteria as well as the assimilative capacity of the receiving stream under different flow conditions. The flow variable WLAs can be calculated by the following equation:

$$FVM = (C_s - C_r) * 5.39 * 1 * MZ \text{ or } ZID \qquad (14.0\text{-}1)$$

Where:

FVM = Acute or chronic flow variable mass WLA, lbs/day/cfs
C_s = Acute or chronic water quality criterion, mg/l
C_r = Upstream background pollutant concentration, mg/l
5.39 = Unit conversion factor, (8.34 lbs/gallon * 0.646 mgd/cfs)
1 = Per unit stream flow, cfs
MZ or ZID = Fraction of stream flow in the MZ or ZID, 0.25 and 0.025

The flow variable WLAs are converted to Average and Maximum effluent permit limits using the current permit derivation procedure. There will be 12 different flow variable limits, one for each month of the year. It should be noted that for certain parameters, including ammonia nitrogen and $CBOD_5$, the calculated flow variable limits must also meet the dissolved oxygen standard based on the water quality modeling described in $CBOD_5$ and DO WLAs Section.

The following equation is to be used by facilities, at the frequency specified in the permit, to calculate the daily flow variable mass loading:

$$\frac{Q_d * C_d * 8.34}{Q_r} = \text{Flow Variable Mass Loading (lbs/day/cfs)} \qquad (14.0\text{-}2)$$

Where:

Q_r = River Flow, cfs
Q_d = Discharge Flow, mgd
C_d = Pollutant concentration in discharge, mg/l

A facility with flow variable limits will need to monitor and report values for each of the factors in Equation 14.0-2. It is important to note that the Discharge Monitoring Report (DMR) will treat the Flow Variable Mass as any other parameter. Both the monthly average and the daily maximum value will be included in the DMR. Compliance is determined by comparing the monthly average and daily maximum values to the water quality-based flow variable limits derived based on Equation 14.0-1.

15.0 Site-Specific Data Collection

A facility has the option to collect site-specific ambient and effluent water chemistry data in lieu of using statewide default values and submit the data for the department's consideration. Iowa Code 455B.194 (Credible Data Law) requires that the department shall use credible data when establishing a total maximum daily load for any discharge to a Water of the State.

Wastewater treatment facilities are encouraged to plan ahead when considering any local or regional data gathering effort. It is recommended that facilities contact the department's Wasteload Allocation Staff to approve the data collection plan and ensure the results will be applicable to the WLA calculations.

Once the facility collects and then submits the data to the department, staff reviews the data and determines if the data are complete and applicable to the WLA calculations. If the data are not acceptable, then default values for the WLA are used.

15.1. Water Chemistry Data

Site-specific ambient and effluent water chemistry data may be used in lieu of the statewide default values. For example, a facility may collect site-specific effluent pH and temperature, as well as upstream background pH and temperature data, to derive site-specific ammonia limits. Also, site-specific effluent and in-stream background hardness, sulfate, and chloride data may be used to determine site-specific chloride and sulfate limits.

Considerations involved in using local and/or regional data are discussed below:

(1) *Local Values:* Submit a minimum of two years of water chemistry readings sampled at least once per month to establish representative annual site-specific data.
   a. More frequent monitoring in a shorter time period than two years is optional if the facility is able to demonstrate that the monitoring data are collected in a representative year. The factors the department will consider to determine if the data are collected in a representative year include the following:
      1. Stream flow. No drought or flooding conditions; weather patterns typical of the year.
      2. In some cases, for certain pollutants seasonal water chemistry data obtained at the most critical conditions, such as low stream flow and high temperature conditions may be acceptable.
   b. For site-specific *effluent* water chemistry each sample must be a 24-hour composite sample of the final effluent. For intermittent discharges where 24-hour composite sampling is not feasible, a representative grab sample is acceptable.

c. Sampling must not be impacted by rainfall/runoff events. At least 72 hours are recommended between rainfall/runoff period and sampling.
d. Collect data in the receiving stream that is governing the ammonia nitrogen WLA. Contact the department staff for the appropriate stream location.
e. The ammonia nitrogen criteria are monthly values. Site-specific monthly pH and temperature for the effluent and the receiving waterbody, as well as background ammonia values are needed to establish site-specific monthly limits. More frequent monitoring data are needed to establish site-specific monthly water chemistry data for ammonia nitrogen (pH and temperature) WLAs. Therefore, for a parameter that requires monthly site-specific values sample both effluent and stream background at least four times per month (or once per week) for a 24-month timeframe (48 data points) to establish representative site-specific data. Continuous effluent and background temperature data collection is preferred, where possible.
f. Choose a sample location far enough upstream from the wastewater treatment facility outfall to be beyond any potential effluent impacts to the receiving stream.
g. Include the date, time, and quantitative result for each sample collected.

(2) *Regional Background Values:* Determine if regional water chemistry data are available that represents the upstream background conditions. Some examples of where this might be applicable:
a. Another facility, at a reasonable distance upstream (on the same stream) from the facility of interest, has collected background readings of water chemistry data (such as hardness, sulfate, chloride, etc.) that are also representative of the background chemistry for the facility of interest;
b. Ambient monitoring data are available within the same watershed (such as STORET data or USGS data) that are representative of the upstream background conditions of the facility of interest.

Factors that may potentially influence the acceptability of the regional background data include the following:

- The distance from the monitoring location to the outfall;
- Another discharge located between the regional station and the outfall; and/or
- Another tributary flows into the waterbody between the regional station and the outfall.

It should be noted that each situation will be evaluated on a case-by-case basis to determine if the available regional background data are representative of the background values for the discharge of interest.

15.2. Site-Specific Coefficient of Variation Data Collection

The coefficient of variation (CV) is a standard statistical measure of the relative variation of a set of data, and it is defined as "the standard deviation divided by the mean". Iowa's statistical method of permit limit derivation includes estimate determination of the CV for the distribution of the sample measurements of the parameters (such as ammonia nitrogen and toxics) after the facility complies with the WQS. This CV is based on the individual treatment facility's operations. Where the CV data are lacking, a default value of 0.6 is used for ammonia nitrogen and toxics. EPA recommends (U.S. EPA, 1991) a value of 0.6 as a default CV if the regulatory authority does not have more accurate information on the CV for the pollutant parameter. The value of 0.6 is typical of the range of variability of effluents measured by EPA and represents a reasonable degree of relative variability. However, wherever possible, it is recommended that data on effluent variability for the pollutant of concern be collected to define a CV rather than using the default value.

The following describes the specific steps to follow in order to collect site-specific CV values:

(1) Collect effluent data for a specific parameter at a time when the facility is operating normally; operating as designed; and meeting current NPDES permit limits;
(2) Collect a minimum of two years of samples at least once per month in order to reasonably quantify the CV and reduce uncertainty;
(3) If the current NPDES permit has specific sampling/testing monitoring requirements ensure that the site-specific data are collected in a consistent manner.
(4) Collect a 24-hour composite sample of the final effluent for use in effluent water chemistry determinations. For intermittent discharges, when a 24-hour composite sample is not feasible, a representative grab sample is also acceptable;
(5) Include the date, time, and quantitative result of each sample collected.

16.0 CBOD$_5$ and DO WLAs

The calculation of a WLA for conventional pollutants considers the in-stream dissolved oxygen (DO) impacts of carbonaceous biochemical oxygen demand (CBOD), ammonia nitrogen, and any other oxygen demanding materials. The WLA for ammonia nitrogen and other oxygen demanding materials is also addressed in separate sections, as these pollutants can cause acute and chronic toxicity.

The WLAs for oxygen demanding pollutants are determined directly from the results of water quality models, which account for the fate of the pollutants as they move down the receiving stream. The default inputs to the model outlined in the following subsections are used unless other scientifically defensible values are available.

16.1. Water Quality Modeling

The department uses water quality models to predict the effects of point source discharges on DO levels in a waterbody.

Water quality models vary in complexity from simple relationships that model a few processes under specific conditions to very complex relationships which attempt to model many processes under a wide range of conditions. The more simple models use limited information about the system being modeled and are also limited in their applicability. Steady-state models in which certain relationships are assumed to be independent of time fall into this category. More complex models relate many natural processes to several water quality parameters on a time-dependent basis. These models use extensive information about the system being modeled and also have a broader range of applicability. Dynamic models fall into this category.

DO is an example of a parameter where models are used to provide important information. The ability of a stream to maintain an acceptable DO concentration is an important consideration in determining its capacity to assimilate wastewater discharges. DO is used in the microbial oxidation of organic and certain inorganic matter present in wastewater. Oxygen, supplied principally by re-aeration from the atmosphere, replaces DO lost through oxidation processes. When the rate of oxygen consumption exceeds the rate of re-aeration, the DO concentration may decrease to levels below the minimum allowable standards.

To predict the variation in DO, as well as in ammonia nitrogen and $CBOD_5$ concentrations in streams, both a simplified Excel spreadsheet implementing the modified Streeter-Phelps DO equation and a more complex mathematical model such as QUALIIK are used in Iowa. Each model is discussed in further detail in the following sections. Input data for the models are developed from existing technical information and site-specific field investigations of selected streams. When sufficient data are not available, literature data and conservative assumptions are applied until site-specific information becomes available.

16.2. Theory and Methodology

DO concentrations in streams are controlled by many factors including atmospheric re-aeration, biochemical oxygen demand (both, carbonaceous and nitrogenous), algal photosynthesis and respiration, benthal oxygen demands, temperature, and the physical characteristics of the stream. Many of these factors are difficult, if not impossible, to accurately assess. Limitations on the use of these controlling factors are discussed below.

Photosynthesis can produce large quantities of oxygen during the day if algae are present in the stream. Conversely, at night, algal respiration creates an oxygen demand. Both photosynthesis and respiration are included in the QUALIIK model. Phytoplankton photosynthesis is a function of temperature, nutrients, and light. Phytoplankton respiration is represented as a first-order rate that is attenuated at low oxygen concentrations. Benthal oxygen demands result from anaerobic decomposition of settled organic material at the bottom of the stream. These reactions release carbonaceous and nitrogenous organics that create biochemical oxygen demand. The inclusion of benthal oxygen demand in the QUALIIK model requires extensive field surveys to determine the aerial extent of sludge deposits within a stream and coefficients that describe the release into the water. In most instances, data are not available to accurately describe sludge deposition areas. QUALIIK includes the sediment oxygen demand component. The sediment-water fluxes of DO and nutrients are simulated internally rather than prescribed. That is, sediment oxygen demand and nutrient fluxes are simulated as a function of settling particulate organic matter, reactions within the sediments, and the concentrations of soluble forms in the overlying water. The sediment oxygen demand simulation is best used when sufficient field data are available to calibrate and verify the rate constants. If field data are not available, default rate constant values are used.

Nitrogenous biochemical oxygen demand (NBOD) occurs due to the oxidation of ammonia nitrogen to nitrates by certain species of bacteria. This oxidation process is called nitrification. Nitrification is a two-step process whereby a specific bacterial species oxidizes ammonia nitrogen to nitrite, and a different bacterial species oxidizes the nitrite to nitrate. Theoretically, approximately 4.5 mg/L of oxygen are required to oxidize 1.0 mg/L of ammonia nitrogen to nitrate. This theoretical value may conservatively overestimate the oxygen demand of nitrification as the nitrifiers obtain oxygen from inorganic carbon sources during combined energy and synthesis reactions. Actual values obtained have varied between 3.8 and 4.5 mg/L of oxygen per mg/L of ammonia nitrogen. The spreadsheet implementing the Streeter-Phelps equation uses 4.33 as the ratio of NBOD to ammonia nitrogen.

16.3. Modified Streeter-Phelps DO Model

The spreadsheet uses the modified Streeter-Phelps equation to predict DO deficit within the stream. This approach recognizes CBOD and NBOD, atmospheric re-aeration, and initial DO deficit. The effects of photosynthesis and sediment oxygen demands are not specifically considered. The DO mass balance equation is shown as follows:

$$D(t) = \frac{K_d * L_0}{K_r - K_d}\left(e^{-K_d(t)} - e^{-K_r(t)}\right) + \frac{K_N * N_0}{K_r - K_N}\left(e^{-K_N(t-t_0)} - e^{-K_r(t-t_0)}\right) + D_0 e^{-K_r(t)} + \frac{(R-P)}{K_r}\left(1 - e^{-K_r(t)}\right) + \frac{SOD}{K_r * H}(1 - e^{-K_r(t)}) \tag{16.3-1}$$

Where:

D(t) = DO deficit at time t, mg/L

D_o = Initial DO deficit, mg/L
L_o = Initial ultimate carbonaceous BOD concentration, mg/L
N_o = Initial ultimate nitrogenous BOD concentration, mg/L
K_d = Carbonaceous de-oxygenation rate constant, base *e*, day^{-1}
K_N = Nitrogenous de-oxygenation rate constant, base *e*, day^{-1}
K_r = Re-aeration rate constant, base *e*, day^{-1}
T = Time of travel through reach, day
SOD = Sediment oxygen demand, g O_2/ft^2/day
H = Average stream depth, ft
R = Algal respiration oxygen utilization, mg/L/day
P = Photosynthetic oxygen production, mg/L/day
t_0 = Nitrogenous lag time, days

NOTE: For t_0, when a wastewater contains both carbonaceous and nitrogenous oxygen demand, there is a time lag before the onset of nitrogenous oxygen demand. The value of t_0 may be experimentally determined where effluent or stream field measurements are practicable. In the case of well-nitrified effluents, the value of t_0 may generally be considered to be less than 1 day. Note that for t values less than $t_{0,}$ the nitrogenous term does not enter into the calculation of D(t).

Since the initial ultimate NBOD is normally not readily available, it is estimated based on the equation as follows:

$$N_0 = 4.33 * N_{n_0} \qquad (16.3\text{-}2)$$

Where:
N_{n_0} = Initial ammonia nitrogen concentration, mg/L

The ultimate carbonaceous and nitrogenous BOD concentrations as a function of time (t) are calculated as follows:

$$L(t) = L_0 * e^{-K_d(t)} \qquad (16.3\text{-}3)$$

$$N(t) = N_0 * e^{-K_N(t)} \qquad (16.3\text{-}4)$$

Where:
$L(t)$ = Ultimate carbonaceous BOD at time, t, mg/L
$N(t)$ = Ultimate nitrogenous BOD at time, t, mg/L

In the two above equations, the rates of oxygen utilization due to carbonaceous and nitrogenous BOD are expressed as first-order reaction rates. This is an accepted procedure for the carbonaceous demand, but represents a simplification for the nitrogenous demand.

Since nitrification is a two-step process, many researchers have proposed that it is a second order reaction. However, most water quality models use a first order reaction for the ease of programming and usage.

Nitrifying bacteria are generally present in relatively small numbers in untreated wastewater. The growth rate at 20°C is such that the organisms do not exert an appreciable oxygen demand until about eight to ten days have elapsed in laboratory situations. This lag period, however, is reduced or eliminated in a stream for a number of reasons including the discharge of large amounts of secondary effluent containing seed organisms and nitrifier population buildup on the stream's wetted perimeter. In biological treatment systems, substantial nitrification takes place resulting in the build-up of nitrifying organisms. These nitrifying bacteria immediately begin to oxidize the ammonia nitrogen present and exert a significant oxygen demand in a stream below the outfall.

The biological nitrification process is more sensitive to environmental conditions than carbonaceous decomposition. The optimal temperature range for the growth and reproduction of nitrifying bacteria is from 29°C to 33°C. NBOD assumes greatest importance in small streams, which receive relatively large volumes of secondary wastewater during the low flow, warm weather periods of the year (July, August and September). During winter low flow periods (January and February) nitrification will have limited influence upon the oxygen demand due to the intolerance of nitrifying bacteria to low temperatures. During analysis of winter low flow conditions, limited nitrification is observed.

16.3.1. De-Oxygenation Rate Constants

The CBOD decomposition rate in laboratory bottle tests provides a first estimate of the removal rate in natural waters. The CBOD decay rate in the laboratory depends on the degree of treatment of the sewage prior to discharge. The higher treatment levels are corresponding to lower CBOD decay rates.

The CBOD decay rates in the laboratory are rarely directly applicable to surface waters due to the differences in the environment. In fact, only in deep, slow flowing rivers would the CBOD decay rates determined in the laboratory be comparable. In most other rivers, environmental factors tend to make the actual removal higher than for the laboratory bottle rates. The primary causes of this increase are settling and stream bed effects. CBOD settling does not exert an oxygen demand.

Many factors are known to influence the CBOD decay rate including temperature, hydraulic parameters, and degree of wastewater treatment. The temperature effects are discussed in the *Temperature Corrections* section below. Different studies have been conducted to correlate the CBOD decay rate with stream hydraulic parameters.

One model presents the CBOD decay rate as a function of the laboratory decay rate and the stream's hydraulic characteristics using the following equation (Wang and Pereira, 1985):

$$K_d = K_{d_0} + b * \frac{V}{d} \qquad (16.3\text{-}5)$$

Where:
K_d = In-stream CBOD decay rate, at 20 $^{\circ}$C base *e*, day^{-1}
K_{d_0} = Laboratory CBOD decay rate, at 20 $^{\circ}$C base *e*, day^{-1}
b = Bed activity coefficient
V = Flow velocity (fps)
d = Stream flow depth (ft)

The bed activity coefficient, *b*, can be estimated as a step function of stream slope, as shown in Table 16.3-1.

Table 16.3 -1. *Bed Activity Coefficient as a Function of Stream Slope*

Stream slope (ft/mile)	b
2.5	0.10
5.0	0.15
10	0.25
25	0.40
50	0.60

Equation 16.3-5 is used to estimate the CBOD decay rate unless site-specific field data or calibration data are available. The default laboratory CBOD decay rate is 0.2/day.

16.3.2. Nitrification Rate Constant

Information on nitrogenous deoxygenation rates is extremely limited; however, available information indicates that nitrification rates (when active nitrification does occur) are somewhat greater than lab carbonaceous oxidation rates. Therefore, the nitrogenous deoxygenation rate (K_N) (0.3/day at 20°C was selected) is used as input data unless calibration/verification efforts provide a more reliable value.

U.S EPA [2] (1985) has also summarized K_N rates measured in the field and used as parameter values for models from a number of investigations. The K_N rates ranged from 0.15 to 9.0 per day. Field measurements can result in the overestimation of K_N where significant algal or attached periphyton effects occur. Algae consume ammonia nitrogen as a nutrient; therefore, a K_N determination based only on the loss of ammonia nitrogen would include uptake of ammonia nitrogen by algae as well as ammonia nitrogen oxidation.

16.3.3. Re-Aeration Rate Constant

Four re-aeration rate constant estimation methods are provided in the CBOD5/DO modeling. Each re-aeration model is only applicable under certain conditions. Melching and Flores (1999) is the most recent study and includes equations for Pool-Riffle and Channel Control streams based on data collected in streams across the United States. Thus, for small streams with pools and riffles, the Pool-Riffle model is selected. For relatively uniform channels, the Channel Control model is selected. For flows and depths greater than the upper range of data used by Melching and Flores (1999), Bennett and Rathbun's (1972) 2nd Equation or O'Connor and Dobbins (1958) is used. Specifically, the equations developed by Melching and Flores (1999) are used under the depth range of 0 – 10 feet and 0 – 7,410.4 cfs and Bennett and Rathbun (Equation 2) or O'Connor and Dobbins (1958) Equation are used at depths exceeding 10 feet and flows exceeding 7,410.4 cfs.

Other scientifically defensible re-aeration rate models including reaeration models based on new research data are used on a case-by-case basis.

The following section describes different re-aeration rate constant determination methods.

(1) USGS (Pool-Riffle) Melching and Flores 1999 Model. Two formulations are included in this model, and each is suitable for a certain stream flow range:

When stream flow is less than 0.556 m^3/s (or 19.64 cfs), the formulation is as shown below:

$$K_r = 517 * \frac{(V * S)^{0.524}}{Q^{0.242}} \qquad (16.3\text{-}6)$$

Where:

V = *Stream velocity, m/s*
S = *Streambed slope, m/m*
Q = *Stream flow, m^3/s*

When stream flow is greater than 0.556 m^3/s (or 19.64 cfs), the formulation is as shown below:

$$K_r = 596 * \frac{(V * S)^{0.528}}{Q^{0.136}} \qquad (16.3\text{-}7)$$

(2) USGS (Channel-Control) Melching and Flores 1999 Model. Similarly, two formulations are included in this model, and each is suitable for a certain stream flow range:

When stream flow is less than 0.556 m^3/s (or 19.64 cfs), the formulation is as shown below:

$$K_r = 88 * \frac{(V * S)^{0.313}}{H^{0.353}} \qquad (16.3\text{-}8)$$

Where:

H = *Stream water depth, m*

When stream flow is greater than 0.556 m^3/s (or 19.64 cfs), the formulation is as shown below:

$$K_r = 142 * \frac{(V * S)^{0.333}}{H^{0.66} * B_t^{0.243}} \qquad (16.3\text{-}9)$$

Where:

B_t = *Top width of the channel, m*

(3) Bennett and Rathbun (1972) Equation 2.

$$K_r = 5.58 * \frac{V^{0.607}}{H^{0.1.689}} \qquad (16.3\text{-}10)$$

(4) O'Connor and Dobbins (1958) Equation.

$$K_r = 3.93 * \frac{V^{0.5}}{H^{1.5}} \qquad (16.3\text{-}11)$$

The above reaeration formulas all give values K_r that approach zero as the depth of the stream increases, implying that reaeration becomes negligible for deep bodies of water. This is certainly not the case, since when water motion is less significant, wind becomes the dominating factor in reaeration. The reaeration constant typically has a minimum value in the range shown below:

$$K_r = \frac{0.6}{H} \; to \; \frac{1.0}{H} \qquad (16.3\text{-}12)$$

If the calculated value of K_r falls below the range of minimum values given in Equation 16.3-12, K_r = 0.6/H is used.

16.3.4. Temperature Corrections

Temperature corrections for the carbonaceous de-oxygenation rate constant, nitrification rate constant, and the re-aeration rate constant are performed within the computer model. The following equations define the specific temperature corrections used in the program:

$$K_d(T) = K_d(20) * (1.047^{(T-20)}) \qquad (16.3\text{-}13)$$

$$K_r(T) = K_r(20) * (1.024^{(T-20)}) \qquad (16.3\text{-}14)$$

$$K_N(T) = K_N(20) * (1.083^{(T-20)}) \qquad (16.3\text{-}15)$$

Where:

T = *Water temperature, ℃*

The temperature corrections for the three rate constants are commonly used (U.S. EPA [2], 1985).

16.3.5. DO Saturation

The principal factor affecting the solubility of oxygen is the water temperature, pressure and salinity (or conductivity). DO saturation values at various temperatures, salinity (or conductivity) and pressure are calculated based on the following formulas as described in Anderson & Huggins (2003), and Chapra (1997).

Temperature Effect:

$$C_s = e^{-139.34411+\frac{1.575701\times 10^5}{T+273.15}-\frac{6.642308\times 10^7}{(T+273.15)^2}+\frac{1.243800\times 10^{10}}{(T+273.15)^3}-\frac{8.621949\times 10^{11}}{(T+273.15)^4}} \qquad (16.3\text{-}16)$$

Where:
T = Water temperature, °C
C_s = Saturation value for DO in fresh water, at temperature T at standard pressure of 1 atm, mg/L

Salinity Effect:

The following equation can be used to establish the dependence of saturation on salinity (Chapra, 1997):

$$C_{ss} = e^{ln(C_s) - S\left(1.7674\ x\ 10^{-2} - \frac{10.754}{T+273.15} + \frac{2.1407\ x\ 10^3}{(T+273.15)^2}\right)} \qquad (16.3\text{-}17)$$

Where:
S = Salinity, g/L
C_{ss} = Saturation value for DO in water with salinity S, mg/L

Salinity can be related to chloride concentration by the following approximation:

$$S = 1.80655\ \times\ CL \qquad (16.3\text{-}18)$$

Where:
CL = Chloride concentration, g/L

When conductivity instead of salinity is available, the following steps are used to take into account the salinity effect on DO saturation concentration. The conductivity units must be converted from the units (mS/cm) for conductivity$_1$ to (µS/cm) for conductivity$_2$ using the following equation:

$$Conductivity_2 = Conductivity_1\ \times\ 1000 \qquad (16.3\text{-}19)$$

Then the conductivity correction factor ($\mathrm{f}_{\mathrm{cond}}$) for salinity is computed in the following equation:

$$f_{cond} = -0.000003\ \times\ Conductivity_2 + 1.0002 \qquad (16.3\text{-}20)$$

The correction factor is multiplied by C_s to correct the DO concentration at saturation and standard pressure for salinity as follows:

$$C_{ss} = f_{cond}\ x\ C_s \qquad (16.3\text{-}21)$$

Pressure Effect:

The following equation can be used to establish the dependence of DO saturation on pressure.

$$C_{sp} = C_s \times P \times \left[\frac{\left(1-{}^{P_{wv}}/_{P}\right)(1-\theta P)}{(1-P_{wv})(1-\theta)}\right] \qquad (16.3\text{-}22)$$

Where:

P = $-3 \text{ x } 105 \text{ x Elevation (ft)} + 0.996$

P_{wv} = $e^{11.8571-\frac{3840.7}{T+273.15}-\frac{216{,}961}{(T+273.15)^2}}$

Θ = $9.75 \times 10^4 - (1.426 \times 10^{-0} T) + (6.436 \times 10^{-0}\, T^2)$

16.3.6. Flow Velocity Calculations

Stream flow velocity is important in determining re-aeration rates and the downstream dispersion of pollutants. Site-specific velocity measurements are preferred. Sometimes, the stream velocity is estimated based on stream morphology. When site-specific velocity data are not available, the following default flow velocities are used in the WLA calculation:

a. 0.1 – 0.3 fps in general use streams

b. 0.5 fps in storm sewer or tile line

c. 1 – 2 fps for outfall pipes, pressured pipe flows, such as pressured sewer outfall pipe.

When necessary data are available, the flow velocities may also be estimated based on either a variation of the Manning's Formula for open channel flow or the Leopold-Maddox predictive equation and Hazen-Williams for pressurized pipes.

16.3.7. Manning's Formula

Manning's equation is used where little historical flow and velocity information exists in the stream segment. If flows and velocities are measured during a calibration sampling event, the roughness coefficient "n" is calibrated. However, in most instances, more reliable flow velocity relationships are modeled by using the power equations, discussed in the next section.

Any scientifically defensible approach using Manning's formula is allowable. One approach is to simplify each element in a particular reach to a trapezoidal channel. This approach is discussed below.

Under conditions of steady flow, the Manning equation is used to express the relationship between flow and depth as shown below:

$$V = \frac{1.49 R^{2/3} S^{1/2}}{n} \qquad (16.3\text{-}23)$$

Where:
V = Velocity, fps
R = Hydraulic radius, ft
S = Channel Slope, ft/ft
n = Roughness coefficient

For a river or stream with a width much greater than its depth, the value of R is approximately equal to the mean depth. If both sides of the equation are multiplied by the cross-sectional area of (width) x (mean depth), the following equation results:

$$Q = \frac{1.49 W H^{5/3} S^{1/2}}{n} \qquad (16.3\text{-}24)$$

Where:
H = Mean river depth, ft
Q = Discharge, cfs
W = Water surface width, ft
S = Slope ft/ft
n = Roughness coefficient
A = Flowing area = W*H

All variables except for “H” are input values, thus “H” can be solved for. Velocity, V, can then be calculated by the following formula:

$$V = Q/A = Q/(WH) \qquad (16.3\text{-}25)$$

River slopes are estimated from USGS topographic maps and the department’s LiDAR data. River widths are estimated from field observations of a Use Attainability Analysis (UAA) or USGS field measurements at each USGS gaging station or from other sources of site-specific data.

The following table shows the roughness coefficient for various open channel surfaces.

Table 16.3-2. *The Manning roughness coefficient for various open channel surfaces (from Chow et al., 1988).*

MATERIAL	n
Man-made channels	
Concrete	0.012
Gravel bottom with sides	
Concrete	0.020
mortared stone	0.023
Riprap	0.033
Natural stream channels	
Clean, straight	0.025-0.04
Clean, winding and some weeds	0.03-0.05
Weeds and pools, winding	0.05
Mountain streams with boulders	0.04-0.10
Heavy brush, timber	0.05-0.20

Manning's *n* typically varies with flow and depth. As the depth decreases during periods of low flow the relative roughness usually increases. Typical published values of Manning's *n*, which range from about 0.015 for smooth channels to about 0.15 for rough natural channels, are representative of conditions when the flow is at the bankfull capacity. At critical low flow conditions, the relative roughness is much higher.

In developing the particular model run for a stream segment, depth and velocity data from stream gaging stations or from field surveys are used to extrapolate depth and velocity at other points along the segment. The extrapolation is a rough approximation; however, it is reasonably close over the total length of a stream reach. When available, the results of field investigations to determine actual stream velocities and depths at many selected stream sites in the modeled segment are used.

16.3.8. Power Equations (Leopold-Maddox Relationships)

Power equations (sometimes called Leopold-Maddox relationships) are used to relate mean velocity and depth to flow for the elements in a reach:

$$V = aQ^b \qquad (16.3\text{-}26)$$

$$H = \alpha Q^{\beta} \tag{16.3-27}$$

Where a, b, α, and β are empirical coefficients determined from velocity-discharge and stage-discharge rating curves. The values of velocity and depth are employed to determine the cross-sectional area and width by:

$$A_c = \frac{Q}{V} \tag{16.3-28}$$

$$W = \frac{A_c}{H} \tag{16.3-29}$$

Where:

V = Stream velocity, ft/sec
Q = Discharge, cfs
H = Mean river depth, ft
W = Water surface width, ft
A_c = Cross sectional area, ft^2

It is significant to point out the empirical constants *a* and *b* apply to a specific stream cross section. The value of "*a*" represents the velocity at a unit discharge. The value of "*b*" represents the slope of a logarithmic plot of velocity versus discharge. The exponents, *b* and β, typically take on values listed in Table 16.3-3. Note that the sum of *b* and β must be less than or equal to 1. If this is not the case, the width will decrease with increasing flow. If their sum equals 1, the channel is rectangular.

Table 16.3-3. *Typical values for the exponents of rating curves used to determine velocity and depth from flow (Barnwell et al. 1989).*

Equations	Exponent	Typical Value	Range
$V = aQ^b$	b	0.43	0.40 - 0.60
$H = \alpha Q^{\beta}$	β	0.45	0.30 - 0.50

The power equations are used in many studies and are found to produce reliable results when the empirical constants are properly evaluated. However, their use is limited to streams for which historical data are not available to determine representative values for the empirical constants. When site-specific stream data are available, a regression analysis is performed on several sets of velocity-discharge data to determine the empirical constants. The data selected for use in the analysis usually corresponds to low stream flow conditions since the use of elevated stream flow data may bias the results. When site-specific data are not available, default empirical constants shown in Table 16.3-3 are used.

Reaches of uniform cross section, slope, and roughness parameters rarely characterize stream systems. The same values of the empirical constants usually do not apply to all reaches along a stream segment unless field measured data indicates otherwise. Velocity and discharge values are obtained from the USGS gaging station data or from stream surveys.

16.4. QUALIIK Model

QUALIIK is a river and stream water quality model intended to represent a modernized version of the QUAL2E model (Brown and Barnwell, 1987).

A detailed documentation and User's Manual for the QUALIIK water quality model can be found on the EPA website. The User's Manual provides documentation of the theoretical aspects of the model as well as a description of the model input and data requirements, which are not reproduced in this document. Specific input sequences and formats are presented in the User's Manual. Detailed procedures for calibrating the rate constants to specific stream conditions are also presented in the User's Manual.

16.5. Modeling Data Sources for Models Such as QUALIIK

The bulk of the work in stream water quality modeling is the collection and interpretation of all available data describing the stream system to be modeled. This section describes procedures and data sources that are used in stream modeling for wasteload allocations.

a. Wastewater Discharges

The data for each facility consists of effluent flow rates and effluent characteristics such as biochemical oxygen demand (BOD), ammonia nitrogen, and DO concentrations as well as temperature. Most wastewater discharge information is available in the department's files including WLA requests, NPDES permit applications, as well as facility plans and construction permits.

b. *Mapping Modeled Reaches*

The first step in modeling a river system is determining the locations of all tributaries, wastewater facilities, dams, and other critical points along the river. The total length of the main channel of the river to be modeled is established and river miles are mapped such that the locations of tributaries, wastewater facilities, dams, etc., are identified. Each reach between critical points is then set up as a segment in the model. USGS topographic maps or other maps such as state and county road maps in the department's GIS library are used to supplement the USGS maps.

c. *Field Reconnaissance/Special Stream Surveys*

The following field data is collected during special stream surveys:

(1) The precise location of wastewater discharges.
(2) The location, condition, height, and type of dams and the nature and approximate length of the pool created by the dam.
(3) Approximate river widths at bridge crossings.
(4) Approximate shape of channel cross sections.
(5) Channel characteristics that will aid in determining the channel roughness coefficients.

The special stream survey is performed during flow conditions that represent the flows used in the modeling effort. Stream discharge information during stream surveys are verified from data obtained from the USGS. The stream flow observed during stream surveys is often greater than the 7Q10. Data such as river widths are extrapolated downward to represent 7Q10 conditions. Shapes of channel cross sections are an aid in this determination.

d. *River Channel Slopes*

After river reaches and locations are established, the next step is the determination of river channel slopes. During low flow conditions, it is assumed that river channel slopes are essentially the same as the slope of the water surface unless site-specific data is available. In some cases, profiles of the river have already been determined. The U.S. Army Corps of Engineers does this as part of the work conducted prior to proposal or construction of flood control reservoirs. Without accurate profiles river slopes are determined from USGS contour maps or LiDAR data by locating the points where contour lines cross the river. Stream slopes that are calculated from contour maps only represent an average value over the distance of the river between contour intervals. A GIS elevation coverage is also used to obtain the stream slopes. Often, these are the only sources available and are the best method of slope determination when an extensive field survey has not been completed.

e. River Widths and Roughness Coefficients

River widths and roughness coefficients are estimated using available field data. Roughness coefficients are also estimated using Table 16.3-2 values.

The variation of river widths with discharges is determined from data at USGS gaging stations. The USGS periodically calibrates each gage. The results from these calibrations are available on the USGS website and include widths, cross-sectional areas, mean velocities, and discharges. Reasonably accurate estimations of river widths at the desired discharge are made with this gaging station information along the river widths measured during stream surveys.

f. Stream Flow

In the determination of flow conditions throughout the river system to be modeled, all available data from USGS flow measuring stations if they are available as well as flow rates from all of the wastewater discharges are obtained. River flows are allocated among tributary, groundwater, and wastewater inflow sources. The design low stream flow is used as the modeling basis, and is determined based on the procedures in the Discharge Flow Determination Section.

A summation of tributary inflows and wastewater discharges are sometimes less than the stream flow. The difference is distributed along the main channel of the river as a uniform inflow in terms of cfs per mile of river reach length. If the gaged flow is less than the summation of tributary and wastewater inflows then it is possible to allot a uniform outflow from the main river channel.

g. Tributary and Groundwater Quality

Values for BOD, ammonia nitrogen, and DO levels in tributaries and groundwater inflow are used for stream modeling. If the tributary is free of continuously-discharging wastewater facilities, water quality is assumed to be good. The default tributary water quality input values are as follows:

- Ultimate BOD = 6 mg/L;
- Ammonia nitrogen concentrations = statewide background concentrations
- DO = 6 mg/L

Groundwater is also noted to be of high quality. The model input values for groundwater are ultimate BOD of 6 mg/L and ammonia nitrogen at 0 mg/L. A groundwater DO of 2 mg/L is used in WLA work in Iowa based on USGS groundwater monitoring data.

Facilities have the option to submit site-specific stream or groundwater background chemical concentrations for consideration in lieu of the statewide background concentrations. For more information on site-specific data collection, refer to Site-Specific Data Collection Section of this document.

h. Rate Constants

The re-aeration rate constant (K_r) is determined from one of many available predictive formulas shown above. The document titled "Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling – EPA/600/3-85/040, June 1985" is one source for obtaining the initial values for rate constants.

Carbonaceous and nitrogenous de-oxygenation rate constants are best determined experimentally for a specific wastewater effluent and/or calibrated for a specific stream. However, when specific values are not available, "typical" values from similar streams and default values described in the previous sections are used. Specific explanations of these rate constants are in the User's Manual for the QUALIIK model.

17.0 Permit Derivation Procedure

This section describes the methods used to translate a WLA into an NPDES permit limit. The procedures are applied to any facility in the state (municipal, industrial, or semi-public) for whom a water quality-based effluent limit is required. The purpose of these procedures is to provide an effluent limit that will statistically assure the WQS will not be exceeded due to the variations in facility operation, monitoring, and parameter analysis.

17.1. Maximum Daily Limits (MDL) and Average Monthly Limits (AML)

Maximum Daily Limits (MDL) and Average Monthly Limits (AML) are calculated using the statistical procedure that adopts the 1991 *EPA Technical Support Document* (TSD) methodology. For toxics, this procedure considers the required sampling frequency for each water quality based parameter noted in 567 IAC Chapter 63 and any known coefficient of variation (CV) for each parameter. This CV is based on the individual treatment facility's operations. Where the CV data is lacking, a default value of 0.6 is used. If a wastewater treatment facility opts to increase its monitoring frequency, the corresponding permit limits are calculated to reflect this increased frequency. For ammonia nitrogen, the permit limits are derived directly from the acute and chronic WLAs.

Definition of Variables:

WLA_a = Acute Wasteload Allocation
WLA_c = Chronic Wasteload Allocation
CV = Coefficient of Variation

n = Sampling Frequency
LTA_a = Acute Long Term Average
LTA_c = Chronic Long Term Average
MDL = Maximum Daily Limit
AML = Average Monthly Limit

Statistical-Based Procedure:

The *Modified 1991 EPA Technical Support Document* (TSD) methodology is adapted for the Iowa statistical-based procedure to derive the permit limits from the WLAs. The following sections describe the different procedures used to derive the permit limits for ammonia nitrogen and toxics.

17.2. Ammonia Nitrogen

$$MDL = WLA_a \tag{17.2-1}$$

$$\text{If } WLA_c < WLA_a,\ AML = WLA_c$$
$$\text{Otherwise, } AML = MDL = WLA_a$$

17.3. Toxics

First, a treatment performance level (LTA and CV) is determined to allow the effluent to meet the WLA requirement. Where two requirements are specified based on different duration periods (i.e., WLA_a and the WLA_c), two different performance levels needed to meet WLA_a and WLA_c are calculated.

The LTA_a is Determined by the Following Equation:

$$LTA_a = WLA_a e^{\left[0.5\sigma^2 - z\sigma\right]} \tag{17.3-1}$$

Where:
σ^2 = $\ln\left(CV^2 + 1\right)$

The LTA_c is Determined by the Following Equation:

For 4-day chronic averaging period (i.e., for toxics)

$$LTA_c = WLA_c e^{\left[0.5\sigma_4^2 - z\sigma_4\right]} \tag{17.3-2}$$

Where:

σ_4^2 = $\ln\left(CV^2/4+1\right)$

The z value for the LTA_a and LTA_c is based on a 0.01-probability basis, i.e., the 99th percentile level, with a value of 2.326. The default CV value is 0.6 unless applicable data is provided by the wastewater treatment facility.

Next, permit limits are derived directly from the corresponding LTA value; in other words, the MDL is calculated from LTA_a and the AML is calculated from the LTA_c.

The MDL is Calculated by the Following Equation:

$$MDL = LTA_a e^{[z\sigma - 0.5\sigma^2]} \qquad (17.3\text{-}3)$$

The z value for MDL is based on a 0.01 probability basis, i.e. the 99th percentile level, with a value of 2.326.

The AML is Calculated Using the Equation:

$$AML = LTA_c e^{[z\sigma_n - 0.5\sigma_n^2]} \qquad (17.3\text{-}4)$$

Where:

σ_n^2 = $\ln\left(CV^2/n+1\right)$

The z value for AML is based on a 0.01-probability basis, i.e., the 99th percentile level, with a value of 2.326. The monitoring frequency (n) follows the requirements noted in 567 IAC Chapter 63. However, the *n* value used to calculate the AML is greater or equal to 4/month to guarantee meeting the criterion.

If the above calculated AML is greater than the MDL, set AML = MDL.

18.0 Antidegradation

Wasteload allocations are calculated to meet water quality standards downstream of the point source discharge. Thus, for new or expanded discharges to Waters of the United States, an antidegradation review shall be performed according to Iowa’s water quality standards and the corresponding implementation procedure.

19.0 Alternative Site-Specific Methodology for Water Quality Based Limits

567 IAC 61.2(5) “c” states that site-specific water quality criteria may be allowed if adequate documentation is provided to show that the proposed criteria will protect all existing or potential uses of the surface water. This Section provides several alternative site-specific options to derive water quality based limits, which are recommended by U.S. EPA. Other alternative site-specific criteria will be reviewed and approved as long as they are scientifically defensible.

The U.S. EPA is responsible for publishing aquatic life ambient water quality criteria in accordance with the provisions of Section 304(a) of the Clean Water Act from time to time in order to reflect the latest scientific knowledge. These criteria are generally based on toxicity test data for a minimum of eight families of aquatic life (U.S. EPA [1], 1985). The 304(a) aquatic life criteria are intended to protect 95 percent of a group of diverse genera, unless a commercially or recreationally important species is very sensitive. States use these EPA guideline criteria as they develop their own specific water quality standards.

U.S. EPA has recognized that site-specific factors related to differences between resident species sensitivity and those in the criteria database, and differences between criteria test water and receiving water characteristics that alter chemical toxicity can dramatically change the chemical concentration that is protective of aquatic life at a specific location (e.g., U.S. EPA[2], 1984). For example, it has long been recognized that it is generally the dissolved form of a chemical (i.e., passing through a 0.45 micron pore size filter) that is most toxic to aquatic life. However, National Pollutant Discharge Elimination System (NPDES) discharge limits are based on the “total” (unfiltered sample) concentration of a chemical (40 CFR 122.45). These phenomena are widely recognized as evidenced by EPA’s publication of default “translators” (U.S. EPA, 1996) to convert dissolved metals criteria into equivalent total metals concentrations for use in setting NPDES discharge limits.

EPA recognizes the highly complex and site-specific interactions that dictate chemical bioavailability and toxicity, and provided means by which to account for them in developing national water quality criteria (U.S. EPA, 1984[2]; U.S. EPA [1], 1985). The most widely applied approach is the use of hardness-dependent metals criteria, although pH-dependent criteria are available for constituents such as ammonia and phenol. As the science and regulatory experience in this area has advanced, various approaches have been used to set site-specific water quality criteria. These range from the long-standing consideration of water hardness’ effect on metals toxicity to the use of empirical dissolved metals translators and more sophisticated modeling approaches such as the Biotic Ligand Model (BLM) for copper.

The approaches and methods listed below are resource and time intensive. These methods are most commonly used on a statewide basis rather than a facility basis. Site-specific water quality criteria are subject to the review and approval of the U.S. Environmental Protection Agency unless the criteria or methodology to derive the site-specific criteria have been approved by the U.S. EPA. However, a facility is not prohibited from pursuing these options. The various approaches to setting site-specific water quality criteria for the protection of aquatic life are summarized below. The appropriate EPA guidance documents must be used to determine the specific steps to be followed for each of these methods. If a facility chooses to pursue one of these approaches, it is recommended that the facility contact the department to develop the specific scope of work and data needs.

19.1. Dissolved Metals Translator

The Dissolved Metals Translator (DMT) approach (U.S. EPA, 1996) is predicated on the concept that it is primarily the dissolved phase of a chemical that is most toxic to aquatic life (exceptions to this exist). "Dissolved" constituents have been operationally defined as those that pass through a 0.45 micron pore size filter (e.g., APHA et al., 1992), and this is the definition used in EPA methodologies (e.g., U.S. EPA, 1992). Although a DMT can be developed by calculation of partition coefficients, the most straightforward approach is by direct measurement (US EP A, 1996). Developing a direct-measurement DMT consists of conducting numerous sampling events and measuring the total and dissolved-phase (0.45 micron filtered) metal concentrations. This generally consists of sampling sufficient to encompass seasonal, operational, and other factors altering the form and concentration of the metal of interest. The ratio of the mean total metal concentration to the mean dissolved metal concentration is used to develop a site-specific translator to replace the default EPA translators (U.S. EPA, 1996) used to set total metal permit limits to reflect the dissolved-metals-based criteria. The specific sampling requirements are described in *The Metals Translator: Guidance for Calculating a Total Recoverable Permit Limit from a Dissolved Criterion* (U.S. EPA, 1996).

Although the DMT approach is generally the simplest and least resource-intensive method to set site-specific metals criteria, the dissolved phase of a chemical is not necessarily toxic. For example, a metal or other chemical associated with dissolved organic carbon (DOC) can pass through a 0.45 micron filter, but may not be toxic because the molecule is too large to cross an organism's cell membrane, and/or it may be strongly chelated by DOC and hence stay in solution and not cross an organism's cell membrane (for example, EDTA chelation is often used to chelate metals and decrease their toxicity in toxicity identification evaluations). In cases where the bioavailability of a metal is not best reflected by dissolved metals data, the use of a Water Effects Ratio (WER) may be more appropriate (see below).

Application of the U.S. EPA DMT approach would allow consideration of site-specific factors altering metals bioavailability in Iowa surface waters.

19.2. Recalculation, Resident Species, Indicator Species

EPA published three methodologies to derive site-specific water quality criteria (U.S. EPA, 2015). Each was designed to consider the effects of interactions between the receiving water and the chemical that altered chemical toxicity relative to that in criteria database toxicity tests, and/or considered differences between resident species and criteria database test species sensitivity to a chemical.

19.2.1. Recalculation Procedure

The Recalculation Procedure is used when the sensitivity of resident species differs from the sensitivity of test organisms used to derive state standards. This allows for the elimination of data for organisms not resident to the site or not expected to occur at the site. If the elimination of data for non-resident species results in the eight-family minimum database requirements (MDR) for criteria derivation to not be met, additional toxicity testing must be conducted. The approach may also require a biological assessment of the receiving stream to determine what criteria database organisms are not “resident to the site” and may not be expected to “occur at the site” as defined in the Recalculation Procedure.

19.2.2. Indicator Species Procedure

The Indicator Species Procedure accounts for changes in chemical toxicity as a result of interactions with receiving stream water (e.g., sorption to suspended solids decreasing toxicity by removing the chemical from the dissolved phase). This approach is used when resident species sensitivity is not different from those in the criteria derivation database. This procedure subsequently became known as the Water Effects Ratio (WER) approach. WER test guidelines were published in 1994 (U.S. EPA[1], 1994).

19.2.3. Resident Species Procedure

The Resident Species Procedure accounts for differences between receiving stream characteristics and test water characteristics that can alter chemical toxicity, and differences between receiving stream organisms and criteria database organism sensitivity to the chemical.

19.2.4. Summary of Alternative Site-Specific Methodologies

The recalculation approach consists of criteria re-derivation after removal of non-resident organisms from the database and addition of relevant data published since the criteria were derived. Absent the need for additional testing to meet MDRs, this is the least resource-intensive approach. The WER approach is more resource-intensive in that toxicity testing of conventional laboratory test organisms must be conducted in both receiving stream ("site" water) and laboratory test water. A ratio between the toxicity of the chemical in site water and laboratory water (e.g., site water LC50 value divided by laboratory water LC50 value) is calculated and used to re-set chemical criteria that are subsequently used in NPDES permit limit derivation. The Resident Species Procedure is the most resource-intensive method because it essentially requires derivation of a WER with resident test species (it is the combination of the indicator species procedure and the resident species procedure).

19.3. Streamlined Copper WER

Reliable and predictable WERs have been developed for copper such that EPA developed a streamlined copper WER methodology (U.S. EPA[2], 2001). The procedure requires limited (relative to a full WER test program) acute toxicity testing of copper in site and laboratory water, and use of existing Acute to Chronic Ratios (ACRs) to develop chronic criteria from acute test data.

19.4. Biotic Ligand Model

EPA has incorporated the Biotic Ligand Model (BLM) as an option for derivation of aquatic life criteria for copper (U.S. EPA, 2007). The BLM, based on research by DiToro et al. (2001), is a computer model that predicts copper toxicity based on known interactions of the copper ion at fish gill surfaces, and establishes copper criteria based on water quality parameters that alter copper toxicity. Site-specific data for various water quality parameters (e.g., pH, calcium, sodium, magnesium, dissolved organic carbon, etc.) are required in order derive site-specific copper criteria. BLMs for other metals could also be used to establish site-specific criteria.

19.5. Other Approaches

While all of the above approaches have merit in some situations, any technically valid approach can be used to account for site-specific factors not considered in derivation of state standards. For example, the Salinity Toxicity Relationship (STR) model (Mount and Gulley, 1992) for predicting the toxicity of a mixture of salt ions could be used as a tool in setting site-specific ion criteria when validated for the effluent and receiving stream matrix. Other modeling and/or in-stream biological assessment approaches can also be used. Validation based on site-specific water quality and/or resident species conditions may be needed in some cases.

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Appendix A: pH Modeling Using Site-Specific Alkalinity and Total Inorganic Carbon

The numerical criteria for certain pollutants such as ammonia nitrogen are a function of pH values. Facilities have the option to collect site-specific water chemistry data and submit the data to the department for consideration. If the site-specific data are acceptable, they are used in the criteria derivation and WLA calculations for these pollutants. The following section describes the procedures used to derive site-specific pH values at the end of the MZ and the ZID that are used to derive the chronic and acute criteria for the pollutants. The equations are also used to derive site-specific effluent pH limits on a case by case basis.

The tendency of natural waters to remain within a relatively narrow band of hydrogen ion activity is due to the presence of buffers that resist pH changes. In many freshwater systems much of the buffering is related to alkalinity and total inorganic carbon. The major chemical species considered to constitute alkalinity are dissolved carbon dioxide, bicarbonate, and carbonate ion, together with the hydrogen and hydroxyl ions. The dissolved inorganic carbon species include carbon dioxide, (CO_2), bicarbonate ion (HCO_3^-), and carbonate ion (CO_3^-). When alkalinity and total inorganic carbon data are available, a more detailed pH modeling is performed.

Facilities have the option to collect alkalinity and total inorganic carbon data for the effluent and upstream background of the receiving stream. When the above site-specific data are available, the pH WLA and pH at the end of the MZ and the ZID is estimated using carbonate system equilibrium relationships.

Basic Concepts and Approach

When carbon dioxide is introduced into an aqueous solution, it combines with water to form carbonic acid. The reaction equations are shown below.

$$CO_2 + H_2O \rightleftharpoons H_2CO_3 \qquad \text{(A-1)}$$

$$H_2CO_3 \rightleftharpoons HCO_3^- + H^+ \qquad \text{(A-2)}$$

$$HCO_3^- \rightleftharpoons CO_3^{2-} + H^+ \qquad \text{(A-3)}$$

$$H_2O \rightleftharpoons H^+ + OH^- \qquad \text{(A-3)}$$

Alkalinity and total inorganic carbon are defined as:

$$Alk = [HCO_3^-] + 2[CO_3^{2-}] + [OH^-] - [H^+] \qquad \text{(A-4)}$$

$$C_T = [H_2CO_3] + [HCO_3^-] + [CO_3^{2-}] \qquad \text{(A-5)}$$

The equilibrium constants for reaction equations from (A-2) to (A-3) are:

$$K1 = \frac{[H^+][\,HCO_3^-]}{H_2CO_3} \qquad \text{(A-6)}$$

$$K2 = \frac{[H^+][\,CO_3^{2-}]}{[\,HCO_3^-]} \qquad \text{(A-7)}$$

$$K_w = [H^+]\,[OH^-] \qquad \text{(A-8)}$$

The equilibrium constants are temperature dependent and the temperature functions are shown below.

$$pK_1 = \frac{3404.71}{Ta} + 0.032786Ta - 14.8435 \qquad \text{(A-9)}$$

$$pK_2 = \frac{2902.39}{Ta} + 0.02379Ta - 6.498 \qquad \text{(A-10)}$$

$$pK_w = \frac{4787.3}{Ta} + 7.1321\log_{10}(Ta) + 0.010365Ta - 22.80 \qquad \text{(A-11)}$$

There are five equations from A-4 to A-8 and five unknowns: $[H_2CO_3], [HCO_3^-], [CO_3^{2-}], [OH^-], and\ [H^+]$.

The solutions for the unknowns are shown below:

$$[H_2CO_3] = F_0C_T \qquad \text{(A-12)}$$

$$[\,HCO_3^-] = F_1C_T \qquad \text{(A-13)}$$

$$[CO_3^{2-}] = F_2C_T \qquad \text{(A-14)}$$

Where F_0, F_1 and F_2 are the fractions of the total inorganic carbon in carbonic acid, bicarbonate, and carbonate, respectively.

$$F_0 = \frac{[H^+]^2}{[H^+]^2 + K_1[H^+] + K_1K_2} \qquad \text{(A-15)}$$

$$F_1 = \frac{K_1[H^+]}{[H^+]^2 + K_1[H^+] + K_1K_2} \qquad \text{(A-16)}$$

$$F_2 = \frac{K_1K_2}{[H^+]^2 + K_1[H^+] + K_1K_2} \qquad \text{(A-17)}$$

The relationship among $[H^+]$, C_T and alkalinity are as follows:

$$F_1 C_T + 2F_2 C_T + \frac{Kw}{[H^+]} - [H^+] - Alk = 0 \qquad \text{(A-18)}$$

Thus, the equation C_T is as follows:

$$C_T = \frac{Alk - \frac{Kw}{[H^+]} + [H^+]}{F_1 + 2F_2} \qquad \text{(A-19)}$$

The fourth-order polynomial equation in $[H^+]$ is as follows:

$$\begin{aligned}[H^+]^4 + (K_1 + Alk)[H^+]^3 + (K_1K_2 + Alk * K_1 - K_w - K_1C_T)[H^+]^2 + (Alk * K_1K_2 - K_1K_w \\ - 2K_1K_2C_T)[H^+] - K_1K_2K_w = 0\end{aligned} \qquad \text{(A-20)}$$

Equation A-20 is solved by using Quartic Equation Solver for different alkalinity and total inorganic carbon concentrations that are listed in Table A-1.

Please note all units for the above equations are in moles/L. However, the commonly used unit for alkalinity is mg/L as $CaCO_3$. The unit conversion from mg/L as $CaCO_3$ to moles/L is as follows:

$$Alkalinity\ in\ \frac{Moles}{L} = \frac{X\frac{mg}{L}\ Alkalinity\ as\ CaCO3}{50 * 1000} \qquad \text{(A-21)}$$

Mass Balance Equations for the Mixing Zone and Zone of Initial Dilution

Data Requirements

Q_e = Effluent flow, (cfs)
Q_r = Corresponding Stream flow, (cfs)
$[Alk]_e$ = Effluent alkalinity, mg/L as $CaCO_3$
$[Alk]_r$ = Receiving waterbody alkalinity, mg/L as $CaCO_3$
pH_e = Effluent pH, standard unit
pH_r = Receiving waterbody pH, standard unit
$[CT]_e$ = Effluent total inorganic carbon, mmole/L
$[CT]_r$ = Receiving waterbody total inorganic carbon, mmoles/L
MZ = Mixing zone dilution, dimensionless, between 0 to 1
ZID = Zone of initial dilution, dimensionless, between 0 to 1
pH (WQS) = pH criteria (6.5 to 9.0)

Mass Balance Equations

Steps:

1. Determine $[C_T]_e$ using Equation (A-19) based on effluent alkalinity and pH
2. Determine $[C_T]_r$ using Equation (A-19) based on receiving waterbody alkalinity and pH
3. Calculate $[C_T]$MZ or $[C_T]$ZID in the MZ and ZID

$$[C_T]_{MZ} = \frac{\{[C_T]_e * Q_e + [C_T]_r * MZ * Q_r\}}{(Q_e + MZ * Q_r)} \qquad \text{(A-22)}$$

$$[C_T]_{ZID} = \frac{\{[C_T]_e * Q_e + [C_T]_r * ZID * Q_r\}}{(Q_e + ZID * Q_r)} \qquad \text{(A-23)}$$

Calculate alkalinity at the Mixing Zone and Zone of Initial Dilution

$$[Alk]_{MZ} = \frac{\{[Alk]_e * Q_e + [Alk]_r * MZ * Q_r\}}{(Q_e + MZ * Q_r)} \qquad \text{(A-24)}$$

$$[Alk]_{ZID} = \frac{\{[Alk]_e * Q_e + [Alk]_r * ZID * Q_r\}}{(Q_e + ZID * Q_r)} \qquad \text{(A-25)}$$

Determine pH for the Mixing Zone and Zone of Initial Dilution:

The pH in the ZID can be calculated based on the calculated total inorganic carbon and alkalinity above using either Table A-1 or Quartic Equation Solver for Equation (A-20). The MZ pH equals the site-specific receiving water pH_r based on 567 IAC 61.2(4) “f”.

Table A-1. *pH as Function of Ct and Alkalinity*
(Table values are pH values)

		Values of Ct in the MZ or ZID (mmoles/L at 25 degrees C)																
		2.0	2.5	3.0	3.5	4.0	4.5	5.0	5.5	6.0	6.5	7.0	7.5	8.0	8.5	9.0	9.5	10.0
MZ or ZID Alkalinity (mg/L as CaCO3)	200	10.7	10.4	10.0	9.5	8.3	7.3	7.0	6.8	6.7	6.6	6.5	6.4	6.4	6.3	6.3	6.2	6.2
	210	10.8	10.5	10.1	9.7	9.1	7.5	7.1	6.9	6.7	6.6	6.5	6.5	6.4	6.4	6.3	6.3	6.2
	220	10.9	10.5	10.2	9.8	9.4	7.9	7.2	7.0	6.8	6.7	6.6	6.5	6.5	6.4	6.3	6.3	6.3
	230	11.0	10.6	10.3	9.9	9.6	8.7	7.4	7.1	6.9	6.7	6.6	6.6	6.5	6.4	6.4	6.3	6.3
	240	11.0	10.7	10.4	10.0	9.7	9.2	7.7	7.2	7.0	6.8	6.7	6.6	6.5	6.5	6.4	6.4	6.3
	250	11.1	10.8	10.5	10.1	9.8	9.4	8.3	7.4	7.1	6.9	6.8	6.7	6.6	6.5	6.5	6.4	6.4
	260	11.2	10.9	10.5	10.2	9.9	9.6	9.0	7.6	7.2	7.0	6.8	6.7	6.6	6.6	6.5	6.4	6.4
	270	11.2	10.9	10.6	10.3	10.0	9.7	9.3	8.0	7.3	7.1	6.9	6.8	6.7	6.6	6.5	6.5	6.4
	280	11.3	11.0	10.7	10.4	10.1	9.8	9.5	8.7	7.5	7.2	7.0	6.8	6.7	6.7	6.6	6.5	6.5
	290	11.3	11.1	10.8	10.5	10.2	9.9	9.6	9.1	7.8	7.3	7.0	6.9	6.8	6.7	6.6	6.6	6.5
	300	11.3	11.1	10.8	10.5	10.3	10.0	9.7	9.3	8.3	7.4	7.1	7.0	6.8	6.7	6.7	6.6	6.5
	310	11.4	11.2	10.9	10.6	10.3	10.1	9.8	9.5	8.9	7.7	7.3	7.0	6.9	6.8	6.7	6.6	6.6
	320	11.4	11.2	11.0	10.7	10.4	10.1	9.9	9.6	9.2	8.0	7.4	7.1	7.0	6.8	6.8	6.7	6.6
	330	11.4	11.3	11.0	10.8	10.5	10.2	10.0	9.7	9.4	8.6	7.6	7.2	7.0	6.9	6.8	6.7	6.7
	340	11.5	11.3	11.1	10.8	10.5	10.3	10.0	9.8	9.5	9.0	7.8	7.3	7.1	7.0	6.9	6.8	6.7
	350	11.5	11.3	11.1	10.9	10.6	10.3	10.1	9.9	9.6	9.3	8.3	7.5	7.2	7.0	6.9	6.8	6.7
	360	11.5	11.4	11.2	10.9	10.7	10.4	10.2	10.0	9.7	9.4	8.8	7.7	7.3	7.1	7.0	6.9	6.8
	370	11.5	11.4	11.2	11.0	10.7	10.5	10.2	10.0	9.8	9.5	9.1	8.1	7.4	7.2	7.0	6.9	6.8
	380	11.6	11.4	11.3	11.1	10.8	10.5	10.3	10.1	9.9	9.6	9.3	8.6	7.6	7.3	7.1	7.0	6.9
	390	11.6	11.5	11.3	11.1	10.9	10.6	10.4	10.1	9.9	9.7	9.4	9.0	7.9	7.4	7.2	7.0	6.9
	400	11.6	11.5	11.4	11.2	10.9	10.7	10.4	10.2	10.0	9.8	9.5	9.2	8.3	7.6	7.3	7.1	7.0