Iowa Wasteload Allocation (WLA) Procedure

Iowa Department of Natural Resources
Water Quality Bureau

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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 that are necessary for the protection of surface water quality standards as described in IAC 567 Chapter 61 – Water Quality Standards. A Wasteload Allocation (WLA) is the portion of a receiving water's total maximum daily load that is allocated to one of its existing or future point sources of pollution.

The WLA Procedure document addresses the water quality-based effluent limits for point source discharges requiring a National Pollutant Discharge Elimination System (NPDES) permit such as industrial and sewage treatment plant discharges. IAC 567 Chapter 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 IAC 567 Chapter 64.4(1) *"e"*.

The Iowa Department of Natural Resources (hereinafter referred to as "DNR") is responsible for maintaining and enhancing water quality in the state. To that end, DNR 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 is 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 WLA Procedures 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, bacteria, chloride, sulfate, temperature, and pH. The implementation of general water quality criteria, site-specific data collection requirements and mixing zone procedures are also discussed. Section 14 shows the biochemical oxygen demand (BOD_5) and dissolved oxygen (DO) modeling in order to meet the dissolved oxygen criteria in the downstream waters of the discharge. Using water quality modeling, effluent limits are derived for: BOD_5 or carbonaceous biochemical oxygen demand ($CBOD_5$), ammonia-nitrogen (NH_3-N), and dissolved oxygen.

The WLA Procedure document also includes the permit derivation procedure that describes the methods used to translate a WLA into a water quality based NPDES permit limit. The following Sections describe each topic in details.

2.0 Discharge Flow Determination

Wasteload allocations are calculated for wastewater treatment facilities or for other permitted and regulated activities that may discharge into waterways in order to assure applicable state Water Quality Standards (WQS) are being met in the downstream route of the discharge flow. For continuous wastewater discharges requiring construction permits, wasteload allocation 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 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 IAC 567- 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.

> ***Average Dry Weather (ADW) Flow:*** The daily average flow when the groundwater is at or near normal and runoff is not occurring. The period of measurement and reporting for this flow should extend for up to 30 days.
>
> ***Average Wet Weather (AWW) Flow:*** The daily average flow for the wettest 30 days for mechanical plants or the wettest 180 consecutive days for controlled discharge lagoons. The respective wettest consecutive 30 and 180-day flows may or may not coincide with precipitation events.

For industrial discharges with no wastewater design flows, the ADW and AWW flows are the highest 30-day average discharge flow and the daily maximum discharge flow, respectively. The maximum monthly average flow and daily maximum flow are provided in the NPDES permit application. The use of a maximum monthly average discharge flow and a maximum daily discharge flow in the WLA calculations for industrial discharges where wastewater design flows are not applicable is consistent with EPA’s Permit Writers’ Manual (EPA-833-K-10-001, September 2010), which states “Permitting authority policy or procedures might specify which flow measurement to use as the critical effluent flow value(s) in various water quality-based permitting calculations.” 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: 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

For toxics, the 7Q10 stream flow is used with the ADW design flow to calculate the chronic concentration WLA; the 1Q10 stream flow is used with ADW design flow to calculate the acute concentration WLA. Additionally, the 7Q10 stream flow is used with the AWW design flow to calculate the chronic mass loading; the 1Q10 stream flow is used with AWW design flow to calculate the acute mass loading. If the facility demonstrates the AWW discharge flow is positively correlated with elevated stream flow, that is, the AWW discharge flow does not occur at critical low flow stream conditions such as 7Q10 and 1Q10, then the chronic mass loading is derived based on the chronic concentration WLA (calculated above using ADW discharge flow and 7Q10 stream flow) and the AWW discharge flow. The acute mass loading is derived based on the acute concentration WLA (calculated above using ADW discharge flow and 1Q10 stream flow) and the AWW discharge flow. For ammonia nitrogen, 7Q10 is replaced with 30Q10 for chronic WLA calculations.

To derive wasteload allocations to protect human health criteria and the MCLs for drinking water uses, the harmonic mean flows are used for carcinogenic chemicals. For non-carcinogens, the stream 30Q5 flows are used to derive the chronic wasteload allocations. When stream 30Q5 flows are not available, stream 7Q10 flows will be used in lieu of 30Q5 flows.

3.1. Design Stream Low Flows at USGS-Gaged Locations

Annual design low stream flows at the United States Geological Survey (USGS)-gaged locations are obtained from the USGS using USGS-approved statistical methods and USGS-approved flow records. The USGS periodically publishes annual design low stream flow values, not monthly low flows. The most recent USGS published stream low flows for USGS gages are used for WLA calculations.

DNR also developed an Excel program using the same Log-Pearson Type III statistical method that USGS uses to estimate annual or monthly critical low flows such as 30Q10, 7Q10, and 1Q10 values at USGS-gaged sites. This program is used to derive site-specific stream design low flows at a USGS gage (in lieu of the USGS published low flows) near a point source discharge using the most current USGS-approved stream flow records upon request on a case-by-case basis.

Facilities also have the option to submit site-specific critical low flow data to DNR for consideration as 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 low stream flows on ungaged stream reaches are determined using the most recent low flow-regression equations developed by USGS for Iowa streams. As of the effective date of this Wasteload Allocation Procedures document, the most current information regarding the low-flow study 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 low stream flows at ungagged locations in the future, the most up-to-date USGS published low stream flows for ungagged locations of interest will be used to derive wasteload allocations.

3.3. Monthly Low Stream Flow Estimation

The USGS low-flow study provides annual or fall seasonal low-flow regression equations. In certain situations, monthly 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 low stream flows on a case-by-case basis. DNR 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. DNR explores the use of monthly critical low flows on a case-by-case basis and considers factors such as:

(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. 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 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 IAC 567 — Chapter 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 before the acute and chronic ammonia criteria are applied. 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 which can in turn 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 Section 13.0 of this document.

4.1. Ambient Background pH, Temperature and Ammonia Nitrogen

The receiving waterbody background pH, temperature, and ammonia nitrogen levels are calculated using the best available monitoring data. The available monitoring data are divided into six different categories based on the waterbody characteristics and designated uses. Tables 4.1-1, 4.1-2, and 4.1-3 present the median values as the default statewide ambient background levels for temperature, pH, and ammonia nitrogen for calculating the ammonia nitrogen WLAs. These default statewide ambient background levels are updated periodically when new ambient monitoring data become available.

Table 4.1-1. *Statewide Median Background Temperature Values for Different Waterbodies (°C)*

Month	Warm Water	Mississippi River		Missouri River	Cold Water	Lakes
		Zone II	Zone III			
Jan.	0.3	0.0	0.1	0.5	4.0	0.5
Feb.	0.1	0.0	0.2	0.5	3.6	0.5
Mar.	1.5	2.0	4.1	4.5	6.0	8.2
Apr.	9.3	10.6	12.2	11.2	9.8	12.5
May	15.0	16.6	17.3	17.1	13.2	17.9
Jun.	19.4	22.6	23.4	23.0	16.7	23.0
Jul.	23.5	26.1	26.6	26.0	18.2	26.5
Aug.	24.3	25.0	26.0	25.8	17.5	26.1
Sep.	20.2	20.4	22.0	21.0	14.7	21.0
Oct.	14.2	12.6	14.1	14.0	10.9	16.0
Nov.	8.0	5.6	7.4	7.0	7.5	7.0
Dec.	0.8	0.1	0.9	1.0	4.2	1.0

Table 4.1-2. *Statewide Median Background pH Values for Different Waterbodies*

Month	Missouri River	Mississippi River	Warm Water	Cold Water	Lakes
	pH (Standard Units)				
Jan.	8.1	8.1	8.1	8.1	8.1
Feb.	8.0	8.0	8.0	8.1	8.0
Mar.	8.1	8.1	8.1	8.1	7.9
Apr.	8.3	8.3	8.3	8.1	8.2
May	8.2	8.2	8.2	8.3	8.4
Jun.	8.2	8.2	8.2	8.0	8.5
Jul.	8.2	8.2	8.2	8.1	8.6
Aug.	8.2	8.2	8.2	8.1	8.5
Sep.	8.3	8.3	8.3	8.1	8.2
Oct.	8.3	8.3	8.3	8.1	8.2
Nov.	8.3	8.3	8.3	8.1	7.6
Dec.	8.3	8.3	8.3	8.2	8.3

Table 4.1-3. *Statewide Median Ambient Background Ammonia Nitrogen Concentration (mg/L)*

Month	Mississippi River	Missouri River	Warm waters	Coldwater	Lakes
Jan.	0.140	0.050	0.020	0.000	0.050
Feb.	0.050	0.050	0.080	0.000	0.050
Mar.	0.145	0.110	0.120	0.000	0.000
Apr.	0.033	0.030	0.030	0.000	0.000
May	0.020	0.020	0.030	0.000	0.020
Jun.	0.060	0.010	0.000	0.000	0.028
Jul.	0.040	0.010	0.000	0.000	0.000
Aug.	0.030	0.010	0.000	0.000	0.000
Sep.	0.030	0.010	0.000	0.000	0.000
Oct.	0.040	0.010	0.000	0.000	0.000
Nov.	0.030	0.020	0.000	0.000	0.000
Dec.	0.000	0.040	0.000	0.000	0.040

The statewide ambient background data are used to derive WLAs. Facilities have the option to submit site-specific data or new monitoring data to DNR for consideration.

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

Statewide effluent pH and temperature values are in place for aerated lagoons, mechanical treatment plants and industrial discharges.

Covered lagoon wastewater treatment technology started operation after the year 2000. The statewide effluent pH and temperature values for covered lagoons are derived based on the monitoring data from six covered lagoon facilities that are in operation in Iowa. Table 4.2-1 shows the statewide median effluent pH and temperature values for different treatment technologies.

Table 4.2-1. *Statewide Effluent pH and Temperature Values for Different Treatment Technologies*

Months	Aerated Lagoons		Mechanical Plants		Industrial Discharges		Covered Lagoons	
	pH	Temperature (°C)	pH	Temperature (°C)	pH	Temperature (°C)	pH	Temperature (°C)
Jan.	7.5	4.50	7.67	12.4	7.9	17.83	7.55	5.70
Feb.	8.0	8.10	7.71	11.3	8.1	17.83	7.65	5.00
Mar.	8.4	8.70	7.69	13.1	8.0	27.67	7.7	7.20
Apr.	8.3	14.6	7.65	16.2	8.2	33.89	7.85	10.7
May	8.5	18.8	7.67	19.3	8.3	35.89	7.80	14.4
Jun.	8.5	22.8	7.7	22.1	8.2	38.67	7.75	19.5
Jul.	8.5	25.3	7.58	24.1	8.2	40.61	7.75	22.1
Aug.	8.6	25.3	7.63	24.4	8.2	39.61	7.65	23.9
Sept.	8.6	22.2	7.62	22.8	8.3	34.5	7.60	20.5
Oct.	8.6	16.6	7.65	20.2	8.2	31.89	7.60	16.0
Nov.	8.6	12.4	7.69	17.1	8.2	29.39	7.70	12.0
Dec.	8.4	8.40	7.64	14.1	8.1	24.67	7.60	7.30

Facilities have the option to submit site-specific effluent data for pH, temperature and ammonia nitrogen to DNR for consideration in lieu of using the statewide default values shown above. Site-specific data collection is discussed in Section 13.0.

4.3. Ammonia Nitrogen Decay Calculations

Ammonia nitrogen is non-conservative in the environment and can be oxidized to nitrite and nitrate. 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. A simpler approach for determining ammonia nitrogen decay is to 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 14.0).

In the presence of nitrifying bacteria, ammonia is oxidized first to nitrite, then to nitrate. The chemical reaction is given in Equation 4.3-1:

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \qquad (4.3\text{-}1)$$

The oxidation reaction is first-order and has the form shown in Equation (4.3-2).

$$N_a = N_{a0}\, e^{(Kn*t)} \qquad (4.3\text{-}2)$$

Where:

N_a	=	ammonia nitrogen remaining at any time, t, mg/L
N_{a0}	=	initial ammonia nitrogen concentration, mg/L
K_N	=	nitrification rate, 0.3/day
t	=	time, days

4.4. 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 IAC 567 - Chapter 61.2(4) "e" for a specific discharger 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 30Q10 flow to the effluent discharge flow.

The flow used in the ammonia nitrogen WLA calculations for the MZ and ZID vary with the of dilution ratio. The discharger is separated into one of three categories based on the river and discharge flows:

(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 DNR for consideration in lieu of the above default MZ and ZID values. MZ study data is collected based on the procedures described in Section 12.0 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 (IAC 567 - Chapter 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:

$$ZID\ pH = -\ LOG\ \{(Q_e * 10^{-pHe} + Q_r * ZID * 10^{-pHr})/(Q_e + Q_r * ZID)\} \qquad (4.4\text{-}1)$$

$$ZID\ TEMPERATURE = [(Q_e * T_e) + (Q_r * ZID * T_r)]/(Q_e + Q_r * ZID) \qquad (4.4\text{-}2)$$

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.

4.5. 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 4.5-1:

$$(C_r * Q_r * MZ\ or\ ZID) + C_e * Q_e = C_s(Q_r * MZ\ or\ ZID + Q_e) \qquad (4.5\text{-}1)$$

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 4.5-1 are compared to the allowable ammonia nitrogen concentrations needed to meet the dissolved oxygen standard which are determined according to the water quality modeling procedures described in Section 14.0. The most stringent acute WLA governs. The final acute and chronic WLAs for ammonia nitrogen are then carried forward to the Permit Derivation Procedure (Section 15.0) 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.

5.0 Toxics (Metals and Other Parameters)

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

5.1. In-Stream Background Chemical Concentrations

Iowa Water Quality Standards have defined numerical criteria for priority pollutants. To properly implement these criteria and calculate WLAs for each point source discharge, background concentrations of the pollutants in Iowa surface waters are established. Two main sources of monitoring data are available. One is the Iowa STORET data; the other is the USGS water quality monitoring data. Arsenic monitoring data on the Missouri River by the Nebraska Department of Environment Quality are also used to derive the arsenic background levels. Information and data analysis will be updated periodically as new ambient monitoring data become available. The final median concentrations for priority pollutants are shown in Table 5.1-1.

Table 5.1-1. *Statewide Background Concentrations of Priority Pollutants in Iowa Surface Waters*

Chemicals	Median (ug/L)	Chemicals	Median (µg/L)
1,1,1-Trichloroethane	0.000	Endosulfan	0.000
1,1,2-Trichloroethane	0.000	Endothall	--
1,1-Dichloroethylene	0.000	Endrin	0.000
1,2,4-Trichlorobenzene	0.000	Ethylbenzene	0.000
1,2-Dichloroethane	0.000	Ethylene dibromide	--
1,2-Dichloropropane	0.000	Fluoride	250
2,3,7,8-TCDD (Dioxin)	--	Lindane	0.000
2,4,5-TP (Silvex)	--	Glyphosate	0.000
2,4-D	0.075	Heptachlor	0.000
3,3-Dichlorobenzidine	0.000	Heptachlor epoxide	0.000
4,4' DDT	--	Hexachlorobenzene	0.000
Alachlor	0.050	Hexachlorocyclopentadiene	0.000
Aldrin	0.000	Lead	0.000
Aluminum	700	Mercury (II)	0.000
Antimony	0.000	Methoxychlor	0.000
Arsenic (III)	0.000	Nickel	0.000
Asbestos	--	Nitrate as N	5,865
Atrazine	0.094	Nitrate+Nitrite as N	5,900

Barium	94.0	Nitrite as N	70.0
Benzene	0.000	o-Dichlorobenzene	0.000
Benzo(a)Pyrene	0.000	Oxamyl (Vydate)	0.000
Beryllium	0.000	para-Dichlorobenzene	--
Bromoform	--	Parathion	0.000
Cadmium	0.000	Pentachlorophenol (PCP)	0.000
Carbofuran	0.000	Phenols	0.000
Carbon Tetrachloride	--	Picloram	0.000
Chlordane	0.000	Polychlorinated Biphenyls (PCBs)	--
Chlorobenzene	0.000	Polynuclear Aromatic Hydrocarbons (PAHs)	--
Chlorodibromomethane	0.000	Selenium	0.000
Chloroform	0.000	Silver	0.000
Chloropyrifos	0.000	Simazine	0.000
Chromium (VI)	0.000	Styrene	0.000
cis-1,2-Dichloroethylene	0.000	Tetrachloroethlyene	0.000
Copper	0.000	Thallium	0.000
Cyanide	0.000	Toluene	0.000
Dalapon	--	Total Residual Chlorine (TRC)	--
Di(2-ethylhexyl)adipate	--	Toxaphene	0.000
Bis(2-ethylhexyl)phthalate	0.000	1,2- trans -Dichloroethylene	0.000
Dibromochloropropane	--	Trichloroethylene (TCE)	0.000
Dichlorobromomethane	0.000	Trihalomethanes (total)	0.000
Dichloromethane	0.000	Vinyl Chloride	0.000
Dieldrin	0.000	Xylenes (Total)	0.000
Dinoseb	0.000	Zinc	0.000
Diquat	--		

Facilities have the option to submit site-specific stream background chemical concentrations for consideration in lieu of the statewide background concentrations. For chemical criteria that are pH dependent, the statewide median pH values from Iowa's ambient monitoring stations are used. Facilities also have the option to submit site-specific effluent and background pH and alkalinity values for consideration for pH dependent chemicals. If the site-specific pH and alkalinity data are acceptable, the site-specific pH value at the boundary of the ZID is calculated using the procedure in Appendix A.

5.2. Mixing Zone and Zone of Initial Dilution for Toxics

The regulatory MZ and ZID are included in IAC 567 - Chapter 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. DNR staff uses the default MZ values as defined IAC 567 - Chapter 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 as they become available.

The Mixing Zone Procedure (Section 12.0) 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.

5.3. 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 IAC 567-Chapter 61.2(4) of the Water Quality Standards, 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 simple mass balance of pollutants shown in Equation 5.3-1 is used to calculate the effluent limits necessary in order meet these boundary conditions.

$$(C_r * Q_r * \text{MZ or ZID}) + C_e * Q_e = C_s(Q_r * \text{MZ or ZID} + Q_e) \qquad (5.3\text{-}1)$$

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 15.0). The final concentration limits are derived using the ADW design flow and the mass limits are derived using the AWW design flow.

6.0 Total Residual Chlorine (TRC)

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. The applicable stream numeric criteria are listed in IAC 567 Chapter 61.3(3) of the Water Quality Standards.

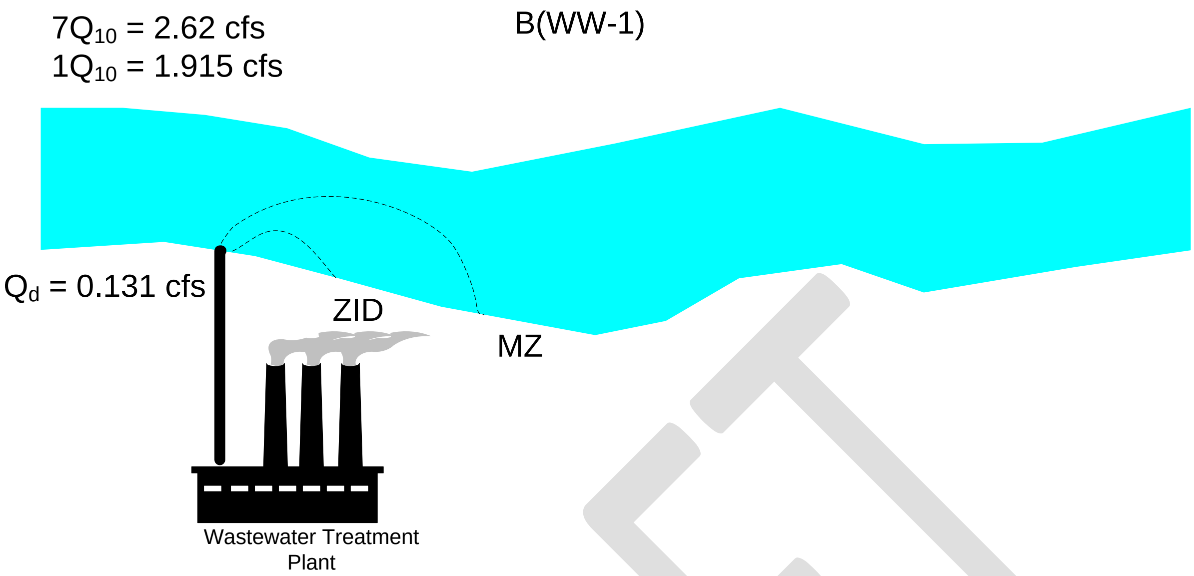
6.1. TRC Wasteload Allocation Calculations

TRC WLAs may be calculated using a combination of mass balance calculations (the same as described for toxic parameters in Section 5.3) 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, 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 (U.S. EPA, 1984; 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 DNR 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.

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 DNR for consideration. The following TRC mass balance equation is used to solve for C_d.

$$C_d*Q_d = C_s*(Q_s + Q_d) + Q_s*X_s + Q_e*X_e \qquad (6.1\text{-}1)$$

Where:

C_r = Background TRC concentration upstream from the outfall, mg/L
Q_r = Design Stream Low Flow (7Q10 for chronic WLA, 1Q10 for acute WLA)
Q_d = Discharge flow, cfs (ADW for concentration limits, AWW for mass limits)
Q_s = (Qr*MZ or ZID) = Stream flow in the regulatory MZ or ZID, cfs
MZ = Stream flow fraction in the regulatory mixing zone (0.25)
ZID = Stream flow fraction in the zone of initial dilution (0.025)
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, 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
Xe = 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 ($WLA_{chronic}$)

$Cr*Qr*MZ + Cd*Qd = C_s(Qr*MZ + Q_d)$
$(0.0)0.655 + C_d(0.131) = 11(0.655 + 0.131)$
$0 + C_d(0.131) = 11(0.786)$
$C_d = 8.646/0.131$
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})

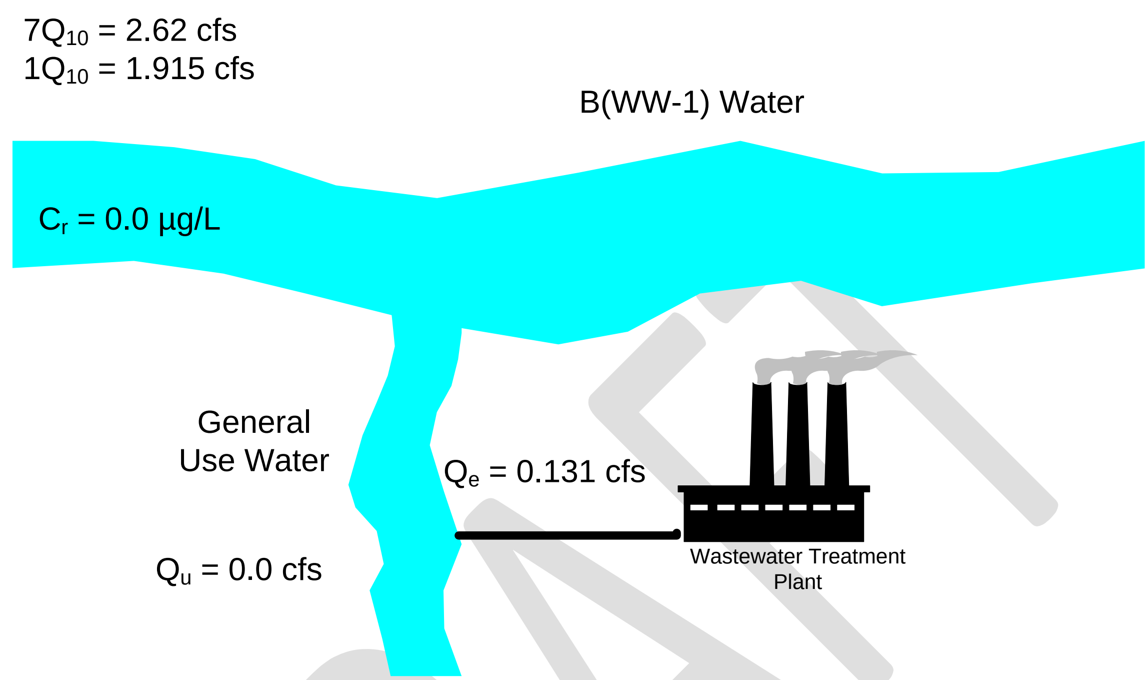
Step 3: Convert WLAs to Permit Limits Using Permit Derivation Procedure (Section 15.0)

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.

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

KEY:
7Q10 = 7-day 10-year low flow
1Q10 = 1-day 10-year low flow
C_r = Background TRC concentration, µg/L
Qr*AMZ = Stream flow in the far field ambient MZ of the designated Class B(WW-1) stream, cfs
Qr*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



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.

$$C_e = C_d * e^{(k*t)} \qquad (6.1\text{-}2)$$

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

$$C_e = C_d e^{(k*t)}$$
$$= 25.95e^{(20)(0.162)}$$
$$= 25.95(25.54)$$
$$C_e = 663 \text{ µg/L} = 0.663 \text{ mg/L } (WLA_{acute})$$

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 (Section 15.0).

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.

7.0 **Bacteria**

IAC 567 - Chapter 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 IAC 567-Chapter 61.3(5) are to be protected for primary contact, secondary contact, and children's recreational uses, respectively. The general criteria of IAC 567-Chapter 61.3(2) and the specific *E. coli* criteria apply to all Class "A" waters. The implementation of the *E. coli* criteria is presented for both continuous and intermittent discharges.

7.1. Continuous Discharges

40 CFR § 122.45(d) states: "For continuous discharges all permit effluent limitations, standards, and prohibitions, including those necessary to achieve water quality standards, shall unless impracticable be stated as:

(1) *Maximum daily and average monthly discharge limitations for all dischargers other than publicly owned treatment works; and*

(2) *Average weekly and average monthly discharge limitations for POTWs."*

The following section describes the methodology used to derive the geometric mean (GM) monthly average and maximum daily *E. coli* limits based on the geometric mean *E. coli* criteria for Class "A" waters.

First, a WQS needs to be converted to a WLA. For bacteria, the criteria apply at the end of pipe. Thus, the WQS for bacteria becomes the WLA. A WLA is then translated to a treatment performance level based on a long-term average (LTA) concentration and coefficient of variation (CV). The water quality-based effluent limits are calculated based on the performance levels to meet the WLAs.

Let Y = ln(X), where X is the untransformed concentration (number of organisms per 100 mL). "ln" is the natural log. The variability of Y (in log scale) in a data set with multiple X's is summarized by a coefficient of variation (CV), where $CV_Y = \sigma_Y/\mu_Y$, where σ_Y represents the standard deviation of the logarithms (of the concentration) and μ_Y is the average of the logarithms (of the concentration).

The LTA concentration (expressed as geometric mean, or $LTA_{Y,m} = \ln(LTA_{X,m})$ where m is the number of days in the averaging period in the permit is consistent with attaining the m-day duration GM criterion specified in the WQS. There is a high degree of assurance that the permitted LTA concentration does not exceed the natural log of the criterion concentration, ln(WQS), where ln(WQS) is equal to the m-day GM (expressed in log space) Therefore, the LTA (expressed in log scale) protects the m-day geometric mean criterion 99% of the time. In other words, the m-day GM criterion is set as the 99th percentile to derive the long-term GM average of $LTA_{Y,m}$ (in log scale):

$$LTA_{y,m} = Ln(WQS)/(1 + Z_{99}\frac{CV_y}{\sqrt{m}}) \qquad (7.1\text{-}1)$$

Z_{99} = 2.326 for 99th percentile occurrence probability. Equation (7.1-1) shows that the LTA is dependent on the duration period of the bacteria criteria. The GM average monthly limit (GM_n) (assuming "n" samples per month) is allowed to vary above the LTA as long as is consistent with meeting the LTA. The Iowa permit derivation procedure uses the 99th percentile probability level:

$$Ln(GM_n) = LTA_{y,m}(1 + Z_{99}\frac{CV_y}{\sqrt{n}}) \qquad (7.1\text{-}2)$$

The relationship (in log space) of the monthly average permit limit to the m-day criterion concentration is thus given by:

$$\frac{Ln(GM_n)}{Ln(WQS)} = (1 + Z_{99}\frac{CV_y}{\sqrt{n}})/(1 + Z_{99}\frac{CV_y}{\sqrt{m}}) \qquad (7.1\text{-}3)$$

The daily maximum limit is allowed to vary above the LTA and it is set as the 99th percentile value:

$$Ln(Max) = LTA_{y,m}(1 + Z_{99}CV_y) \qquad (7.1\text{-}4)$$

The relationship (in log space) of the daily maximum permit limit to the m-day criterion concentration is thus given by:

$$\frac{Ln(Max)}{Ln(WQS)} = (1 + Z_{99}CV_y)/(1 + Z_{99}\frac{CV_y}{\sqrt{m}}) \qquad (7.1\text{-}5)$$

The relationship (in log space) of the daily maximum permit limit to the monthly GM concentration limit is thus given by:

$$\frac{Ln(Max)}{Ln(GM_n)} = (1 + Z_{99}CV_y)/(1 + Z_{99}\frac{CV_y}{\sqrt{n}}) \qquad (7.1\text{-}6)$$

Based on EPA's *Recreational Water Quality Criteria - 2012*, the *E. coli* GM of 126 organisms/100 mL is derived based on a 30-day duration and a risk level of Estimated Illness Rate of 36 per 1,000 primary contact recreators. The following monthly GM and maximum daily limits are derived from the above equations by setting "m" to be 30 days. Table 7.1-1 shows the monthly GM and daily maximum *E. coli* limits for a continuous discharge.

Table 7.1-1. *E. coli Monthly GM and Maximum Daily Limits for Continuous Discharge (Using Default* CV_Y *= 0.19, which is derived based on a mean log standard deviation of 0.4)*

Recreational Uses	Monthly Geometric Mean WQS (org/100 mL)	7-Day Average Geometric Mean Limit (org/100 mL)	7-Day Average Geometric Mean Limit (org/100 mL)	Maximum Daily Limit (org/100 mL)
Class A1 or A3	126	213[A]	356[B]	635
Class A2	630	1,266[A]	2,511[B]	5,434

[A]: 5/week Sampling Frequency
[B]: 2/week Sampling Frequency

7.2. Intermittent discharges

Intermittent discharges such as controlled discharge lagoons do not often discharge 30 days consecutively. The calculation of the GM is difficult due to the lack of adequate number of sample results for each discharge period. Thus, a maximum daily limit or average weekly limits are more appropriate. The average weekly limits or maximum daily limits are based on the same risk levels or same level of protection as the GM value, which is derived for these types of discharges. The average weekly limits or maximum daily limits shown in Table 7.2-1 are applied to intermittent discharges.

Table 7.2-1. *E. coli Monthly GM and Maximum Daily Limits for Non-Continuous Discharges (Using Default CV_y = 0.19, which is derived based on a mean log standard deviation of 0.4)*

Recreational Uses	7-Day Average Geometric Mean Limit (org/100 mL)	Maximum Daily Limit (org/100 mL)
Class A1 or A3	356[A]	635
Class A2	2,511[A]	5,434

[A]: 2/week Sampling Frequency

For wet weather-related events that influence episodic discharges such as combined sewer overflows (CSOs), the 1994 CSO Control Policy (reflected in §402(q) of the CWA) describes various approaches for addressing CSO discharges in NPDES permits and is consulted when establishing water quality based effluent limits for intermittent dischargers.

7.3. *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 traditional relationship 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.

$$C_d = C_e e^{(k*t)} \qquad (7.3\text{-}1)$$

Where:

C_d	=	Allowable *E. coli* discharge concentration, org/100 mL
C_e	=	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 DNR 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 DNR reviewed recent studies on bacteria decay rates (Anderson, K.L., et al., 2005; USGS, 2001-2002; U.S. EPA, 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 1.03/day. Thus, the bacteria decay rate of 1.03/day at 20 °C is used for rivers, streams, lakes, ponds.

8.0 **Chloride and Sulfate**

The chloride and sulfate criteria are included in IAC 567 - Chapter 61.2(4)"b". Both the chloride and sulfate criteria are hardness dependent. Chloride criteria also depend on sulfate concentrations and sulfate criteria are a function of chloride levels. 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 is not available, statewide default water chemistry values are used. The values were determined by analyzing DNR ambient water monitoring data from 2000 to 2007. The statewide default background concentrations are presented below:

- Hardness – 200 mg/L as $CaCO_3$
- Sulfate – 63 mg/L
- Chloride – 34 mg/L

When utilizing the above default background concentrations the water quality criteria for chloride is an acute concentration of 629 mg/L and a chronic concentration of 389 mg/L. For sulfate, the default water quality criterion for aquatic life protection is 1,514 mg/L.

8.2. WLAs for Chloride and Sulfate

When site-specific ambient background and/or effluent water chemistry data are either not available or not adequate to develop site-specific water chemistry values, statewide default water chemistry values are used to calculate the chloride and sulfate criteria as well as the WLAs.

WLAs for point source dischargers in regard to chloride and sulfate are calculated in the same manner as those pollutants listed in Table 1 IAC 567- Chapter 61, Water Quality Standards. 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. 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.

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

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 (Section 15.0).

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:

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

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 standards at the boundary of the MZ default to the statewide background values (hardness of 200 mg/L as $CaCO_3$; sulfate of 63 mg/L; and chloride of 34 mg/L) unless site-specific upstream background water chemistry values are provided by the discharger. 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 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.

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.

9.1. Temperature Criteria

The temperature water quality criteria are included in IAC 567 – Chapter 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, IAC 567 - Chapter 61.2(5)"a" and IAC 567 - Chapter 61.2(5)"b" include the following statements:

a. The allowable 3°C temperature increase criterion for warm water interior streams, IAC 567-Chapter 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 IAC 567-Chapter 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 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.

9.2. Heat Transfer Theory

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

$$H = mc\Delta T \qquad (9.2\text{-}1)$$

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 9.2-2 incorporating the appropriate conversion factor is:

$$H = nQ\Delta T \quad (9.2\text{-}2)$$

Where:

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

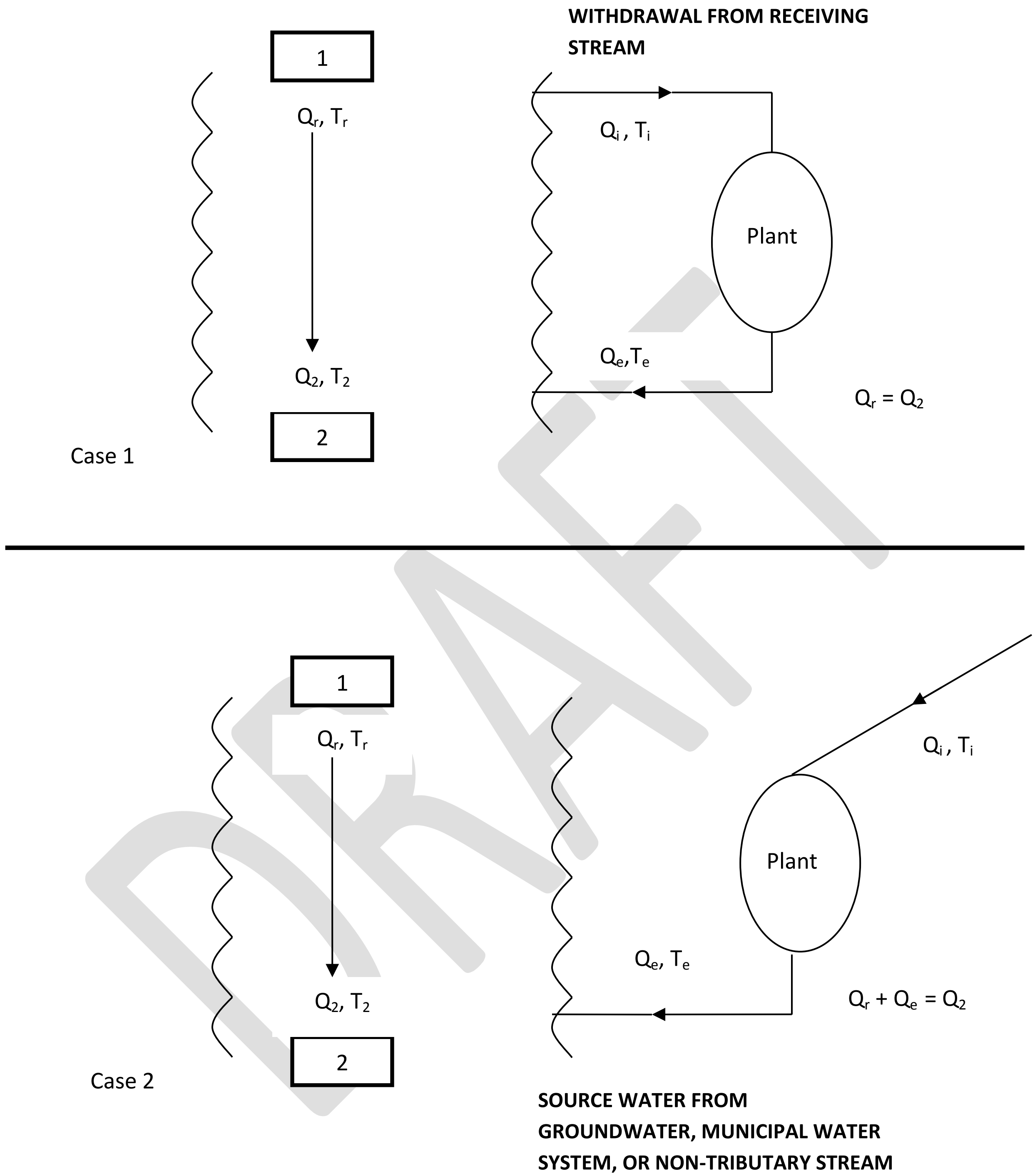
Thus, the amount of heat gained or lost is determined from the heat transfer equation:

$$(Q\Delta T)\text{ gained} = (Q\Delta T)\text{ lost} \quad (9.2\text{-}3)$$

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 9.2-1. 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 9.2-1



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

9.3. 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 ΔT↑ criteria are as follows:

$$T_{max} \text{ calculation: } (Q_e + Q_r{*}MZ){*}T_{max} = Q_r * MZ * T_r + Q_e{*}\ T_e \qquad (9.3\text{-}1)$$

$$\Delta T\uparrow \text{ calculation: } T_e = T_r + 3°C * (Q_e + Q_r{*}MZ)/Q_e \qquad (9.3\text{-}2)$$

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), shown in Tables 9.3-1, 9.3-2, 9.3-3, and 9.3-5
MZ	=	Allowable mixing zone fraction, dimensionless, 0-1
ΔT↑	=	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 fishery, 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 discharger 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 DNR 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 12.0 of this document).

<u>Implementation of IAC 567 – Chapter 61.2(5)”a” and IAC 567 - Chapter 61.2(5)”b”</u>

According to IAC Chapter 61.2(5)”a” and IAC 567 – Chapter 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 such 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 in lieu of 7Q10 is used in the above formulas.

(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.

<u>Warm Water Interior Streams and the Big Sioux River</u>

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

The statewide 90th percentile and maximum monthly background temperatures for streams designated as warm waters are shown in Table 9.3-1. The 90th percentile values 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.

Table 9.3-1 *Statewide 90th Percentile and Maximum Background Temperatures for Streams Designated As Warm Water (IAC 61.3(1)"b")*

Month	90th Percentile Background Temperature (°C)	Maximum Background Temperature (°C)
Jan.	2.3	12.6
Feb.	1.7	17.9
Mar.	6.0	21.8
Apr.	14.6	25.0
May	19.2	26.4
Jun.	23.6	29.5
Jul.	27.4	34.3
Aug.	28.2	33.5
Sep.	24.6	32.7
Oct.	19.8	31.0
Nov.	12.0	18.0
Dec.	5.4	13.0

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 9.3-3:

$$T_{e\text{-average}} = T_r + 3°C * (Q_e + Q_r * MZ)/Q_e \qquad (9.3\text{-}3)$$

Where:

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

T_r = Ambient <u>maximum</u> background temperature (°C), shown in Table 9.3-1

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 9.3-4:

$$T_{e\text{-}max} = T_r + (32\ °C - T_r) * (Q_e + Q_r * MZ)/Q_e \qquad (9.3\text{-}4)$$

Where:

$T_{e\text{-}max}$	=	Maximum allowable effluent temperature, (°C)
T_r	=	Ambient 90th Percentile background temperature (°C), shown in Table 9.3-1

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 would need to 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.

<u>Cold Water Streams</u>:

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

(1) Criteria: IAC 567 - Chapter 61.3(3)"b"(5)"2" states "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 (9.3-5) and (9.3-6).

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

$$T_{e\text{-}average} = T_r + 2°C * (Q_e + Q_r * MZ)/Q_e \qquad (9.3\text{-}5)$$

Where:

$T_{e\text{-}average}$	=	Allowable Average effluent temperature, (°C)
T_r	=	Ambient <u>maximum</u> background temperature (°C), shown in Table 9.3-2

$$T_{e\text{-}max} = T_r + (20\,°C - T_r) * (Q_e + Q_r * MZ)/Q_e \qquad (9.3\text{-}6)$$

Where:

$T_{e\text{-}max}$	=	Maximum allowable effluent temperature, (°C)
T_r	=	Ambient <u>90th Percentile</u> background temperature (°C), shown in Table 9.3-2

(3) Background temperature values: The statewide background temperature values derived for cold water streams are shown in Table 9.3-2.

Table 9.3-2: *Statewide 90th Percentile and Maximum Background Temperature for Streams Designated As Cold Water (IAC 61.3(1)"b")*

Month	90th Percentile Background Temperature (°C)	Maximum Background Temperature (°C)
Jan.	6.4	7.3
Feb.	7.0	10.0
Mar.	9.5	14.0
Apr.	13.1	18.0
May	16.6	19.7
Jun.	20.7	27.1
Jul.	21.8	27.8
Aug.	21.2	28.9
Sep.	18.7	24.6
Oct.	13.6	18.3
Nov.	9.1	10.1
Dec.	7.3	8.1

The Missouri River

The procedures for calculating temperature limits for discharges to the Missouri River are the same as for warm water streams except for the following:

(1) Stream design low flow, Q_r: 7Q10 is the allowable stream design low flow for all seasons,
(2) Background temperature values: The statewide background temperature values derived for the Missouri River are shown in Table 9.3-3.

Table 9.3-3: *90th Percentile and Maximum Background Temperatures for the Missouri River*

Month	90th Percentile Background Temperature (°C)	Maximum Background Temperature (°C)
Jan.	2.5	5.0
Feb.	3.0	8.0
Mar.	8.9	16.1
Apr.	15.0	20.0
May	21.2	25.6
Jun.	25.7	28.0
Jul.	28.4	32.0
Aug.	27.8	30.0
Sep.	24.6	30.0
Oct.	18.5	21.5
Nov.	10.8	15.0
Dec.	3.0	9.5

The Mississippi River

Monthly average, daily maximum, and rate of change limits are calculated according to the temperature criteria described in IAC 567 - Chapter 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 9.3-4 during more than 1 percent of the hours in a 12-month period ending with any month. The 90th percentile and the maximum monthly background temperatures for the Mississippi River Zone II (from Iowa north border to Wisconsin – Illinois border) and Zone III (Northern Illinois border to Iowa-Missouri state line) are shown in Table 9.3-5 below. 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 *ΔT↑* 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 9.3-7:

$$T_{e\text{-}average} = T_r + (3°C) * (Q_e + Q_r\ MZ)/Q_e \qquad (9.3\text{-}7)$$

Where:

T_r = Ambient <u>maximum</u> background temperature (°C), shown in Table 9.3-5

Daily Maximum Limits:

WQS state that at no time shall the water temperature exceed the maximum limits in Table 9.3-4 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 9.3-8:

$$T_{e\text{-}max} = T_r + [2°C + (T_{II} \text{ or } T_{III}) - T_r] * (Q_e + Q_r * MZ)/Q_e \qquad (9.3\text{-}8)$$

Where:

T_{II} = T_{cap} criterion for Zone II (°C)

T_{III} = T_{cap} criterion for Zone III (°C)

T_r = Ambient <u>90th Percentile</u> background temperature (°C), shown in Table 9.3-5

Table 9.3-4: *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, 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

Table 9.3-5: *90^{th} Percentile and Maximum Background Temperatures for the Mississippi River*

Month	Zone II		Zone III	
	90^{th} Percentile Temperature (°C)	Maximum Background Temperature (°C)	90^{th} Percentile Temperature (°C)	Maximum Background Temperature (°C)
Jan.	0.9	4.0	1.8	5.1
Feb.	1.2	5.0	2.4	7.2
Mar.	7.8	16.0	9.6	17.7
Apr.	14.5	19.0	15.8	22.0
May	20.3	28.0	21.7	26.9
Jun.	25.8	31.0	26.2	29.4
Jul.	28.3	30.9	29.2	32.4
Aug.	27.5	31.3	28.9	33.5
Sep.	24.1	29.0	25.4	29.5
Oct.	16.7	21.1	18.1	22.3
Nov.	9.7	15.5	11.0	16.0
Dec.	3.0	10.0	4.5	10.0

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

The calculation is described by Equation 9.3-9:

$$T_{e\text{-}max1\%} = T_r + (T_{II\ or\ III} - T_r)*(Q_e + Q_r*MZ)/Q_e \qquad (9.3\text{-}9)$$

Where:

$T_{e\text{-}max1\%}$ = Allowable effluent temperature to meet T_{cap} (°C)

T_r = Ambient 90^{th} Percentile background temperature (°C), shown in Table 9.3-5

Rate of Change (ΔT↑) 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 the 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 for the discharge to acclimate to the decreasing temperature. The decrease in temperature at the end of the calculated mixing zone shall not exceed 1°C per hour."

The effluent temperature limit based on meeting the ΔT↑ 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.

The default ambient background temperature values shown in Tables 9.3-1, 9.3-2, 9.3-3 and 9.3-5 are updated periodically as new ambient monitoring data become available. Each facility has the option to collect site-specific ambient background temperature data and submit the data to Iowa DNR 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.

9.4. 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.

9.4.1. 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 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 as described in Section 3.3.

9.4.2. 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 DNR 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

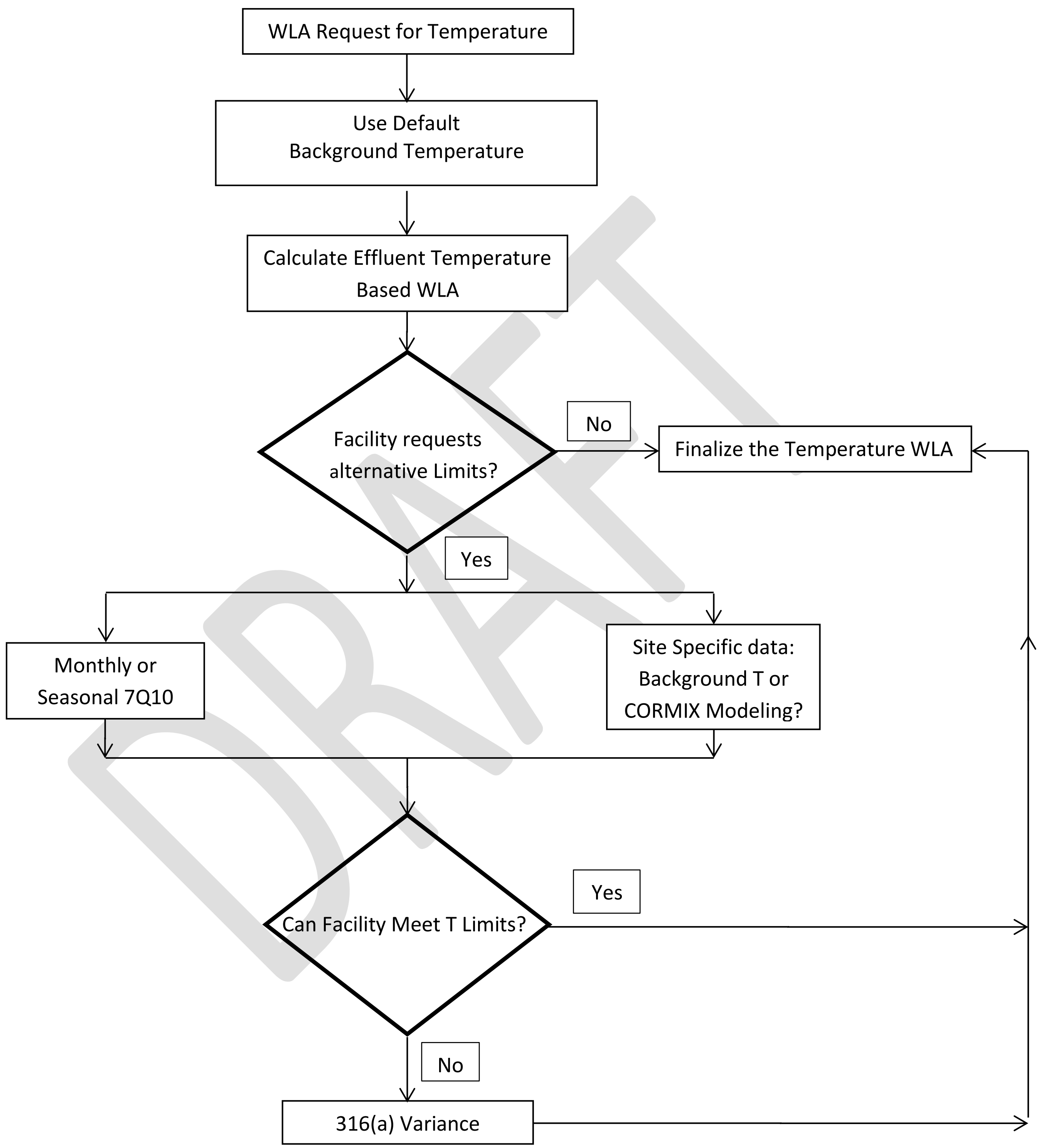
Facilities have the option to collect site specific upstream background temperature to use in lieu of the default background temperatures in the calculation of temperature limits. In order to establish site specific background temperature for each month, adequate monitoring data are required. At least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

9.4.3. 316(a) Demonstrations

Section 316(a) of the Federal Water Pollution Control Act shows that a discharger can be granted an alternate thermal effluent limitation if the discharger 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 discharger with an interest in possible alternate effluent limits based on Section 316(a) should 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 9.4-1 on the next page shows the temperature criteria implementation process.

***Figure 9.4-1** Temperature Criteria Implementation Process Chart*



9.5. Heat loss (temperature "decay") in Pipes and General Use Waters

Iowa's temperature criteria apply to designated waters as specified in IAC 61.3(1)"b"). When effluent temperature limits are calculated for discharges flowing through conveyances such as storm sewers, tile lines, discharge pipes and/or general use segments, the heat losses are considered. Heat is not conservative; and dissipates through convection, conduction, radiation, and evaporation, etc. Heat losses (temperature decay) in pipes are estimated based on convection and conduction heat transfer theory. Heat loss in general use waters is determined using the USGS-developed software, Stream Segment Temperature Model (SSTEMP). Facilities have the option of submitting site-specific heat loss data to the DNR for consideration in lieu of the default heat loss estimations.

10.0 pH

The pH WQS applies to both Class "A" and Class "B" designated waters, and is described in IAC 567 - Chapter 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. The following section explains the methodology.

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

$$pH\,(WQS) = -\log \{(Q_e * 10^{-pHe} + Q_r*MZ*10^{-pHr})/(Q_e + Q_r*MZ)\} \qquad (10.0\text{-}1)$$

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 10.0-1:

$$pH_e = -\log \{((Q_e + Q_r*MZ)*10^{-pH(WQS)} - Q_r*MZ*10^{-pHr})/Q_e\} \qquad (10.0\text{-}2)$$

Equation 10.0-2 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.

11.0 Narrative Water Quality Standards

IAC 567 – Chapter 61.3(2) subsections 'a' – 'h' specifically mention eight general water quality criteria that apply to all surface waters including general use and designated use waters. These criteria are considered in setting the WLA for a discharge to streams, including designated use streams. In waters with no designated use(s), these eight criteria are also applied. 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 narrative criteria as written in IAC 567- Chapter 61.3(2) are incorporated into NPDES permits as standard conditions. In addition, the criterion (IAC 567-Chapter 61.3(2)"d"), which states that waters must be free from of any substance which is acutely toxic to human, animal or plant life, and the protection of livestock watering (IAC 567-Chapter 61.3(2) "g") is discussed in the following section.

11.1. General Use Segments (IAC 567-Chapter 61.3(2) "d")

IAC 567 Chapter 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 based on the acute toxicity end point of LC_{50} (or Effective Concentration, EC_{50}) is the use of ½ of the 96-hour or 48-hour LC_{50} (or EC_{50}) values (U.S. EPA, 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 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 DNR 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 water quality criterion in the following mass balance equation:

$$Cr*Qr + Ce*Qe = NOEC \text{ or } (½ \, LC_{50} \text{ or } ½ \, EC_{50})*(Q_r + Q_e) \qquad (11.1\text{-}1)$$

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 as defined in Section 2.0 , 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 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
MZ	=	Mixing zone dilution, dimensionless, 0-1
C_e	=	WLA concentration for the pollutant of concern, µg/L

Equation 11.1-1 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 15.0). It is important to note the narrative criteria translator value is applied at the end-of-pipe for General Use waters since General Use segments are ephemeral streams with zero design low flows.

11.2. Designated Waters (IAC 567 Chapter (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 11.1-2:

$$C_r*Q_r*ZID + C_e*Q_e = NOEC \text{ or } (1/2 \, LC_{50} \text{ or } ½ \, EC50)*(Q_r*ZID + Q_e) \qquad (11.1\text{-}2)$$

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 as defined in Section 2.0, 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 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 11.1-2 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 15.0).

11.3. Livestock Watering IAC 567 Chapter 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 11.3-1. *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.

12.0 Mixing Zone Procedures

As stated in the *EPA Water Quality Standards Handbook* (U.S. EPA, 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) (USEPA, 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."*

IAC 567-Chapter 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 should 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.

IAC 567 – Chapter 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.* IAC 567-Chapter 61.3(2) *General Water Quality Criteria states "the following criteria are applicable to all surface waters* *including general use and designated use waters, at all places and at all times for the uses described in Subrule 61.3(1)"a".*

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 IAC 567 – Chapter 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 IAC 567 – 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 IAC 567 – Chapter 61.2(4)"a". IAC 567 – Chapter 61.2(4)"d"(4) also allows facilities to collect site-specific MZ data to submit to DNR for consideration. A discharger 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 Section 12.1. Unless site-specific MZ data are submitted for consideration, default regulatory MZs as defined in IAC 567 - Chapter 61 are used.

12.1. Mixing Zone Study Field Procedure Protocol

Facilities have the option of collecting and analyzing site-specific data in a MZ study. Once the facility submits the MZ study to DNR, the DNR reviews the study and determines if the data and analysis is complete and whether it is applicable to the WLA calculations. If the data and analysis is not usable, then default values for the WLA are used.

Wastewater treatment facilities are encouraged to plan ahead when considering any data gathering effort. It is recommended that facilities working on a MZ study contact the DNR Wasteload Allocation Staff to approve the MZ study plan and ensure all data collection and analysis will be applicable to the WLA calculations.

If the facility chooses to perform a dye study, then the discharger is *required* to obtain authorization of the use of biodegradable and nontoxic dye by DNR if the dye has the potential to reach any Waters of the State. The dye's aquatic toxicity and human health effects are to be provided when the request for the use of dye is submitted to DNR. 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 etc. may be used as tracers. The corresponding DNR field office is to be informed of the MZ study by the facility at least 48 hours before the study is conducted.

Basic field data collection options used for MZ studies are described in parts "a" through "c" shown below. Facilities are to submit this field data to DNR Wasteload Allocation staff for calculation of the site-specific MZ. The purpose of the calculation is to reflect the local MZ conditions using site-specific data in lieu of default MZ values.

Stream Characteristics:

NOTE: The terms, "low flow" and "low stream flow," are used in the following discussion. These terms are not synonymous with the "stream design low flow". A facility has the option to provide information on the actual MZ characteristics during low stream flow conditions to demonstrate a different percentage of the low stream flow in the MZ than is projected by the DNR.

Stream surveys to gather MZ data are collected as near to the stream design low flow as is normally feasible during the year. A MZ study is performed at stream flows not exceeding three to five times the stream design low flow. It is recommended that a facility submit a MZ study plan that uses other low stream flow conditions to the DNR Wasteload Allocation staff to ensure the data and analysis are applicable to the WLA calculations. Stream flow conditions closer to the stream design low flow are desirable for those locations where normal flows during the year approach the design low flow or where the flows are controlled by impoundments. Facilities have the option to consider several different field efforts in obtaining the MZ information: Visible Assessment, Dye Injection – Visible Boundary, and Dye Injection – Fluorometric Measurements. These are discussed in greater detail below:

a. Visible Assessment: This procedure is a simple field documentation of the effluent's mixing with the stream flow under low stream flow conditions. Pictures, video, drawings, and/or a detailed map along with physical stream data is provided to illustrate how the two waters (effluent and receiving stream) are combining. Some facilities have added dye to the effluent to facilitate the visible assessment. This approach is adequate on a smaller, shallow stream. A letter from the DNR authorizing the discharge of dye is *required* before dye is introduced into the stream.

   The objective of this assessment is to demonstrate whether or not the effluent flow is completely mixing with the stream within the allowed MZ length. Without additional documentation of the mixing characteristics in the ZID, the default percentage of design low flow defined in IAC 567 – Chapter 61 is used for the ZID in the WLA calculations.

The visible assessment description includes 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 200 feet upstream of the outfall (or other distances that are representative of the upstream background conditions):

(1) Describe the stream bed materials: sand, silt, fine or coarse gravel, or rock;

(2) Note pools and riffles and areas of uniform depths. Estimate length and number thereof and the rapidity of the variations (i.e. gradual, alternating occasionally, or alternating frequently);

(3) Describe the amount of weed growth, snags or obstructions in the stream in terms of: negligible effects on the stream flow characteristics to severe effects on the stream flow characteristics;

(4) Describe the amount of meandering within the 2,000-foot distance; and

(5) Describe other features that might impact the MZ such as delta formation at the stream mouth, islands, other discharges, perennial springs, etc.

(6) Describe the outfall during a low stream flow period. This includes an indication of the discharge flow during the period being described, preferably with pictures. Include such things as the size and configuration of splash pools, outfall height or depth, outfall diameter (if normally filled during discharging), and/or average velocity of flow exiting outfall when submerged.

b. Dye Injection – Visible Boundary Measurements: The objective of this procedure is to provide greater accuracy in characterizing the mixing of an effluent with the receiving stream by using a visible dye injected into the effluent. The following is a brief summary of the procedures that should 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.

(2) Assemble boundary marking floats or stakes. Test stream depth for float line length and ability to wade.

(3) Run short test of dye introduction into the effluent. The dye introduction is normally poured as a slug of dye into the effluent at the last manhole or at the outfall.

(4) Run an actual dye study and set out markers. Obtain time of travel between stations.

(5) Measure stream flow in the dye plume (depth, velocity, cross section) at the boundaries of the MZ and the ZID and total stream flow upstream from the outfall. Obtain effluent flow measurement at time of dye injection.

(6) Prepare a report of the findings.

Specific stream conditions may warrant a modified MZ study plan. It is critical to perform the study at or near design low stream flow conditions. Models are available to project the percentages of mixing obtained during field flow conditions to design low flow regime.

c. Dye Injection - Fluorometric Boundary Measurements: The objective of this procedure is to provide even greater accuracy in characterizing the mixing of an effluent with the receiving stream by using a florescent dye injected into the effluent.

   This study is similar to the Dye Injection – Visible Boundary Measurements effort noted above; however, the actual measurements of dye concentrations (or collection of water samples for later analysis) are made at various locations in the MZ. The dye is fed into the effluent at a constant rate/concentration over the duration of time required to collect all dye samples. The collection of dye samples (or measurement of concentrations) is made across the stream from the shoreline until a point in the stream is reached where additional dye is expected to be present. The same station locations as noted in "b"(1) above are used starting at the lower location and proceeding upstream, and include stream flow measurements as noted in "b"(5) above.

   It is important to note that tracers such as specific conductivity, chloride, and temperature (due to the differences between the levels in the discharge and the ambient water) are an option to use to replace dye products for delineating the MZ boundaries.

Use of MZ Study Results:

The DNR uses the MZ study results to calculate water quality based effluent limits. It is recommended that a MZ study be performed by the facility and submitted to DNR Wasteload Allocation staff prior to NPDES Permit re-issuance. DNR has the ability to provide the facility with preliminary water quality-based permit limits to aid in evaluating the need for a MZ study. DNR Wasteload Allocation staff will review MZ study results to determine if the data and analysis are applicable to the WLA calculations.

12.2. Installation of a Diffuser

An in-stream diffuser to disperse effluent across a more significant portion of the stream is an optional artificial means of increasing the percentage of stream flow contained in the MZ. Typically 50% to 80% of the low stream flow is in the MZ when effluent is discharged via a diffuser, depending on site specific conditions. A construction permit is usually required for a diffuser design and installation. Further requirements regarding the use of an effluent diffuser device are found in IAC 567-Chapter 61.2(4) “d” (3).

12.3. Special Limitations of Mixing Zones

The following conditions define where the use of the default MZ is not appropriate:

(1) Where drinking water contaminants are of concern, MZs shall not encroach on drinking water intakes;

(2) Where MZs and ZIDs are not allowed for bioaccumulating pollutants including Mercury, Chlordane, PCB and Dieldrin;

(3) MZs or ZIDs are not allowed for state-owned lakes and wetlands (which is consistent with IAC 567 - 61.2(4)”d”(1) and 61.2(4)”g”(1);

(5) A ZID is not be allowed in waters designated as Class B(CW), Cold Water (which is consistent with IAC 567 – 61.2(4) “d”(2) and 61.2(4) “g”(2);

(6) MZs or ZIDs are not allowed for reservoirs on streams designated as Class B waters (including Coralville Lake, Lake Red Rock, and Saylorville Lake);

(7) For backwater conditions such as those found on the Mississippi River 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.

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

12.4. Multiple Discharges into the Same Reach

When the discharges to the same waterbody from two or more facilities are in close proximity the development of the WLAs is based on the combined impact on water quality. Step one is to identify the pollutants of concern the facilities have in common. Next the combined ADW flows of the facilities discharging common pollutant(s) are used in deriving the concentration limits. Each facility discharging common pollutant(s) will have the same concentration limits. The combined AWW flows of the facilities discharging common pollutant(s) are used to derive the combined mass loading limits. The overall mass loading limits are then allocated to each facility discharging the common pollutant(s) of concern. There are different mechanisms to allocate the total mass loadings for the combined discharges to each discharger. The total mass loading limits are typically allocated proportionately based on the AWW flows of each facility discharging common pollutant(s) of concern unless site-specific conditions justify the use of an alternative method.

13.0 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 DNR consideration. Iowa Code 455B.194 (Credible Data Law) requires that the DNR 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 DNR 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 DNR, the DNR reviews the data and determines if the data is complete and applicable to the WLA calculations. If the data is not acceptable, then default values for the WLA are used.

13.1. 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 a 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 DNR 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 should 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. No sampling should occur during rainfall/runoff conditions. 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 IDNR 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 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 is 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.

13.2. 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 *E. coli*, 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 is lacking, a default value of 0.6 (except for *E. coli*) is used. 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) Do not sample during rainfall/runoff conditions. At least 72 hours is recommended between a rainfall/runoff period and the collection of samples;
(4) If the current NPDES permit has specific sampling/testing monitoring requirements ensure that the site-specific data is collected in a consistent manner.
(5) 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;
(6) Include the date, time, and quantitative result of each sample collected.

14.0 $CBOD_5$ 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.

14.1. Water Quality Modeling

The DNR 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. If, however, 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.

14.2. 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. Sediment oxygen demand is not included in the Excel spreadsheet model. 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. Assuming secondary wastewater treatment plant effluents contain ammonia nitrogen levels of 10 mg/L during summer operations and 15 mg/L during winter periods, the equivalent NBOD (should all the ammonia nitrogen be converted to nitrates) is approximately 40 to 46 mg/L (summer) and 62 to 68 mg/L (winter).

14.3. 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 Streeter-Phelps equation that is implemented in the spreadsheet is as follows:

$$D(t) = \frac{K_d * L_0}{K_r - K_d}\left(e^{-K_d(t)} - e^{-K_r(t)}\right) + \frac{K_N * N_0}{K_r - K_N}\left(e^{-K_N(t-t_0)} - e^{-K_r(t-t_0)}\right) + D_0 e^{-K_r(t)} + \frac{(R-P)}{K_r}\left(1 - e^{-K_r(t)}\right) + \frac{SOD}{K_r * H}\left(1 - e^{-K_r(t)}\right) \qquad (14.3\text{-}1)$$

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:

$$N_0 = 4.33 * N_{n_0} \qquad (14.3\text{-}2)$$

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:

$$L(t) = L_0 * e^{-K_d(t)} \quad (14.3\text{-}3)$$

$$N(t) = N_0 * e^{-K_N(t)} \quad (14.3\text{-}4)$$

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 26°C to 30°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.

14.3.1. 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 (14.3.4) 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):

$$K_d = K_{d_0} + b * \frac{V}{d} \qquad (14.3\text{-}5)$$

Where:

K_d	=	In-stream CBOD decay rate, at 20 °C base *e*, day^{-1}
K_{d_0}	=	Laboratory CBOD decay rate, at 20 °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 14.3-1.

Table 14.3 -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 14.3-5 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.

14.3.2. 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 (KN) (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 (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.

14.3.3. Re-Aeration Rate Constant

Five re-aeration rate constant estimation methods are provided in the $CBOD_5$/DO modeling spreadsheet. Each re-aeration model is only applicable under certain conditions. USGS Pool-Riffle and Channel Control - Melching and Flores 1999 Models are the most recent models based on stream conditions across United States. Thus, for small streams with pools and riffles, the USGS Pool-Riffle model is usually selected. For relatively uniform channels, the USGS Channel Control model is usually selected. For relatively larger and deeper channels, O'Connor & Dobbins 1958 Model usually provides the best results as long as the model is used within the stream depth and flow velocity conditions. The spreadsheet gives the user the option to choose the most suitable re-aeration model for each specific case. The following section describes different re-aeration rate constant determination methods.

(1) Tsivoglou & Neal 1976 Model. This formulation is based on the premise that the re-aeration capacity of non-tidal fresh water streams is directly related to the energy expended by the flowing water, which in turn is directly related to the change in water surface elevation.

$$K_r = C \times S \times V \tag{14.3-8}$$

Where:

K_r	=	Re-aeration rate constant, base *e*, day^{-1}
S	=	Streambed slope, m/m
V	=	Stream velocity, m/s
C	=	Constant: 31,183 for stream flow between 1 cfs to 15 cfs (0.0283 to 0.4247 m^3/s) 15,308 for stream flow between 15 to 3,000 cfs (0.4247 to 84.95 cms)

(2) Owens et.al. 1964 Model. This is also called the Owens-Gibbs model. This formulation applies for stream velocity in the range of 0.1 to 5.0 feet per second (fps) (0.030 m/s to 1.524 m/s) and stream flow depth in the range of 0.4 to 11 feet (0.122 to 3.35 m).

$$K_r = 5.32 * \frac{V^{0.67}}{H^{1.85}} \tag{14.3-9}$$

Where:

H	=	*Stream water depth, m*

(3) O'Connor & Dobbins 1958 Model. This formulation is more accurate when applied to moderately-deep to deep channel streams. The suitable water channel depth ranges from 1 foot to 30 feet (0.305 to 9.14 m) with a velocity range from 0.5 fps to 1.6 fps (0.152 m/s to 0.488 m/s).

$$K_r = 3.93 * \frac{V^{0.5}}{H^{1.5}} \tag{14.3-10}$$

(4) 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:

$$K_r = 517 * \frac{(V * S)^{0.524}}{Q^{0.242}} \tag{14.3-11}$$

Where:

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:

$$K_r = 596 * \frac{(V * S)^{0.528}}{Q^{0.136}} \quad (14.3\text{-}12)$$

(5) 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:

$$K_r = 88 * \frac{(V * S)^{0.313}}{H^{0.353}} \quad (14.3\text{-}13)$$

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

$$K_r = 142 * \frac{(V * S)^{0.333}}{H^{0.66} * B_t^{0.243}} \quad (14.3\text{-}14)$$

Where:
B_t = *top width of the channel, m*

14.3.4. 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:

$$K_d(T) = K_d(20) * (1.047^{(T-20)}) \quad (14.3\text{-}15)$$

$$K_r(T) = K_r(20) * (1.024^{(T-20)}) \quad (14.3\text{-}16)$$

$$K_N(T) = K_N(20) * (1.083^{(T-20)}) \quad (14.3\text{-}17)$$

Where:
T = *Water temperature, ℃*

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

14.3.5. DO Saturation

The principal factor affecting the solubility of oxygen is the water temperature. DO saturation values at various temperatures are calculated based on the following formula from *Standard Methods for the Examination of Water and Wastewater, 21th Edition*:

$$C_S = \exp(-139.34411 + \frac{1.575701 \times 10^5}{T + 273.15} - \frac{6.642308 \times 10^7}{(T + 273.15)^2} + \frac{1.243800 \times 10^{10}}{(T + 273.15)^3} - \frac{8.621949 \times 10^{11}}{(T + 273.15)^4})$$

(14.3-18)

Where:

T	=	*Water temperature, ℃*
C_s	=	*Saturation value for DO, at temperature T at standard pressure of 1 atm, mg/L*

14.3.6. 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.

14.3.7. Manning's Formula

Each element in a particular reach is idealized as a trapezoidal channel.

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

$$V = \frac{1.49 R^{2/3} S^{1/2}}{n} \qquad (14.3\text{-}18)$$

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 (width) x (mean depth), the following equation results:

$$Q = \frac{1.49WH^{5/3}S^{1/2}}{n} \qquad (14.3\text{-}19)$$

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:

$$V = Q/A = Q/WH \qquad (14.3\text{-}20)$$

River slopes are estimated from USGS topographic maps and DNR 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 14.3-2. *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.

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 below.

14.3.8. 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:

$$V = aQ^b \qquad (14.3\text{-}21)$$

$$H = \alpha Q^\beta \qquad (14.3\text{-}22)$$

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:

$$A_c = \frac{Q}{V} \qquad (14.3\text{-}23)$$

$$W = \frac{A_c}{H} \qquad (14.3\text{-}24)$$

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 14.3-3. 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 14.3-3. *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 14.3-3 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.

14.4. 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.

14.5. 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 discharger 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 DNR 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 dischargers, 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 dischargers, 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 DNR 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 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 14.3-2 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 Section 2.0, Discharge Flow Determination.

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 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.

h. Rate Constants

The re-aeration rate constant (K_r) is determined from one of many available predictive formulas shown in section 14.3.3. 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.

15.0 Permit Derivation Procedure

This section describes the methods used to translate a WLA into an NPDES permit limit. The procedures are applied to any discharger 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 are not be exceeded due to the variations in facility operation, monitoring, and parameter analysis.

15.1. 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 modified *1991 EPA Technical Support Document* (TSD) methodology. For toxics, this procedure considers the required sampling frequency for each water quality based parameter noted in IAC 567 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 describes the different procedures used to derive the permit limits for ammonia nitrogen and toxics.

15.2. Ammonia Nitrogen

$$MDL = WLA_a \qquad (15.1\text{-}1)$$

If $WLA_c < WLA_a$*, AML =* WLA_c

Otherwise, AML = MDL = WLA_a

15.3. 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:*

$$LTA_a = WLA_a e^{\left[0.5\sigma^2 - z\sigma\right]} \qquad (15.3\text{-}1)$$

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)

$$LTA_c = WLA_c e^{\left[0.5\sigma_4^2 - z\sigma_4\right]} \qquad (15.3\text{-}2)$$

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.

<u>The MDL is Calculated by the Following Equation:</u>

$$MDL = LTA_a e^{[z\sigma - 0.5\sigma^2]} \quad (15.3\text{-}3)$$

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.

<u>The AML is Calculated Using the Equation:</u>

$$AML = LTA_c e^{[z\sigma_n - 0.5\sigma_n^2]} \quad (15.3\text{-}4)$$

Where:

$\sigma_n^2 \quad = \quad \ln\left(CV^2/n+1\right)$

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 the IAC 567-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.

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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 DNR 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 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 dependant 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:

$$F_1C_T + 2F_2C_T + \frac{Kw}{[H^+]} - [H^+] - Alk = 0 \qquad \text{(A-18)}$$

Thus, the equation CT 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:

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

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 IAC 567 Chapter 61.2(4)f.

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

MZ or ZID Alkalinity (mg/L as CaCO3)	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
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