11/22/2010

Issue Paper:
Basin Support Document Revisions

1. The Mixing Zone for Implementation of Temperature Standard

IAC 61.3(3)"b"(5) describes the temperature criteria for Iowa waters. The implementation procedure is included in the Supporting Document for Iowa Water Quality Management Plans, Chapter IV (a.k.a Basin Support Document: page 53-54: Thermal Discharges). The current temperature implementation in the Basin Support Document only addresses the 3°C rise condition in the temperature criteria. The revised implementation procedure will include all conditions in the temperature criteria described in IAC 61.3(3)"b"(5).

The mixing zone policy for the temperature criteria implementation is as follows: 25% of the stream Q7-10 flow for interior streams, and 10% of the Q7-10 flow for the Mississippi and the Missouri Rivers. Site-specific mixing zone data may be provided in lieu of the default mixing zone values through either thermal plume modeling such as CORMIX or the field mixing zone study noted in Appendix B of the Basin Support Document.

Special limitations on mixing zone:

The following are proposed conditions where a mixing zone might not be appropriate:

(1) Where drinking water contaminants are a concern, mixing zones shall not encroach on drinking water intakes (**already in the current BSD)**.
(2) Mixing zones would be restricted such that they do not encroach on areas often used for harvesting of stationary species such as shellfish.
(3) Mixing zones and zones of initial dilution would not be appropriate for bioaccumulating pollutants, such as Mercury, Chlordane, PCB and Dieldrin.
(4) Mixing zones should not be located in critical habitat for threatened or endangered species.
(5) Mixing zones would not be appropriate where established mussel beds exist, for example, on the Mississippi River, mixing zones and initial dilutions should not be allowed to encroach known mussel beds.
(6) No mixing zones are allowed for State owned Lakes and Wetlands (**already in the current BSD**).
(7) No zone of initial dilution will be allowed in waters designated as Class B(CW), Cold Water (**already in the BSD**).

2. WLA Procedure for pH

The pH standard applies to both Class A and Class B designated waters and it is described in IAC 61.3(3)"a"(2) and 61.3(3)"b"(2). The current Basin Support Document does not include the implementation procedures for pH.

In wasteload allocations, pH criteria are met at and beyond the mixing zone (MZ). The allowed default MZ Dilution for pH is the same as other toxics listed in Table 1 of Chapter 61 – Water Quality Standards, which is 25% of the $Q_{7\text{-}10}$ flow for interior streams and 10% of the $Q_{7\text{-}10}$ flow for the Mississippi and the Missouri River. Site-specific MZ data may be provided in lieu of the default MZ values through either modeling or a field mixing zone study noted in Appendix B of the Basin Support Document.

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

$$pH\ (WQS) = -\log\ \{(Q_e * 10^{-pHe} + Q_r*MZ*10^{-pHr})/(Q_e + Q_r*MZ)\} \qquad (2.1)$$

where:

- Q_e: effluent flow, AWW is used as the Q_e (cfs)
- Q_r: stream flow, annual $Q_{7\text{-}10}$ is used as the Q_r (cfs)
- pH_e: effluent pH, standard unit
- pH_r: stream flow pH, standard unit,
- MZ: mixing zone Dilution, dimensionless, between 0-1
- pH (WQS): pH criteria (6.5 to 9.0)

Rearrange Equation (2.1):

$$pH_e = -\log\ \{((Q_e + Q_r*MZ)10^{-pH(WQS)} - Q_r*MZ*10^{-pHr})/Q_e\} \qquad (2.2)$$

Equation (2.2) provides the allowed effluent pH values in order to meet the pH criteria in the receiving water of 6.5 to 9.

3. Default Ammonia Nitrogen Decay Rate

Ammonia nitrogen is non-conservative in the environment and it can be oxidized to nitrite and nitrate. The ammonia decay can be accounted for in a wasteload allocation when the effluent flow through a discharge pipe, storm sewer or general use stream before it enters designated stream. When site-specific field data are available, the ammonia nitrogen decay in a general use segment of a stream is usually estimated by water quality modeling such as QUALIIK. A simpler approach for estimating ammonia nitrogen decay is to use a default decay rate of 1 mg/L loss per mile in a general use segment of a stream (within a 2 mile distance), which is an estimate based on past modeling data. When distances exceed two miles, a first-order decay equation will be used with decay rates based on the defaults used in QUAL2K modeling.

4. Aquatic Life Criteria Translator for General Use Segments

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.

For aquatic life protection, acute criteria are applied to general use segments when the constituents have a numeric criteria.

5. Wasteload Allocation Procedures for CDLs

Controlled discharge lagoons are designed to have a hydraulic detention time of at least 180 days and discharge twice per year. The 180-day storage capacity and permitted discharge windows (April 15th through June 15th and September 30th through December 21st) are intended to maximize treatment kinetics during storage periods while avoiding discharges during critical periods of warm weather and ice cover during which low stream flows in combination with limited reaeration capacity or high water temperatures increase the vulnerability of aquatic organisms to pollutants. The wasteload allocations for controlled discharge lagoons treating exclusively domestic wastewater are calculated for conservative pollutants only, such as chloride and sulfate, when it is demonstrated that the controlled discharge lagoon facility is well designed and well operated.

However, if monitoring data show elevated levels of ammonia and/or E. coli, additional monitoring and an examination of the facility’s design and operation may be conducted. When adequate monitoring data show the potential to exceed water quality standards, wasteload allocations for other pollutants such as ammonia nitrogen, E. coli, CBOD5, and dissolved oxygen may be calculated for the discharge.

When a controlled discharge lagoon receives wastewater from a Significant Industrial User (SIU) as defined in 567 IAC 60.2 the wasteload allocation will be calculated for all potential pollutants contributed from the SIU, such as ammonia nitrogen, E. coli, CBOD5, metals, priority pollutants, etc.

Fall seasonal or monthly critical low flows, where available, may be used in the wasteload allocation calculations where appropriate for the pollutants of concern and permitted discharge windows. Wasteload allocation values for each specific pollutant are calculated consistent with the procedures established for continuous discharge facilities using the applicable mixing zone (MZ) and zone of initial dilution (ZID) percentages in association with the appropriate critical low flow values.

In certain cases, a stepwise discharge option may be used. In these cases, the controlled discharge lagoon only discharges when the receiving stream flow is at or above a threshold value. This option requires that the lagoon have enough capacity to store the wastewater during stream flows below the threshold value. It also requires the facility to install a staff gage so that stream flow can be accurately measured when there is no nearby USGS gage stations. The selection of the threshold stream flow value will ensure that the water quality standards are achieved when discharge occurs.

The discharge period for controlled discharge lagoons may be significantly shorter than 30 days and may also occur within two separate consecutive calendar months, making collection, analysis and reporting of a monthly geometric mean impractical or impossible. Thus, sample maximum E. coli limits of 1,073 org./100 mL and 5,365 org./100 mL (they are the 99th percentile values corresponding to the geometric mean value of 126 org/100 and 630 org./100 mL, respectively) are used for Class A1 and A2 designations in lieu of the geometric means of 126 org./100 mL and 630 org./100 mL for Class A1 and A2 designations used for a continuous discharge. The use of 99th percentile (or 99% upper confidence level) is reasonable due to the following reasons:

a. The downstream recreational use is most likely infrequent due to the highly variable hydrological flow
b. The derivation of the sample maximum value is consistent with Iowa's permit derivation procedure. Using 126 org./100 mL as the long term geomean value, the 99th percentile value based on Iowa's permit derivation procedure is 1,073 org/100 mL.
c. It is important to note that the one-sided 99th Confidence Level concept used in EPA's 1986 Bacteria Criteria document is the same as the 99th percentile value.
d. The use of sample maximum fits intermittent discharges more appropriately. As a result, this approach can detect noncompliance quickly and avoid the delay due to waiting until adequate samples are collected.

6. NEW TEMPERATURE WLA PROCEDURE

6.0. DEFINITION OF TERMS USED IN THIS SECTION

c	specific heat (BTU/lb/°F)
D	dilution ratio (Qe + Qr MZ)/Qe, unitless
H_e	facility heat rejection (million BTU/day)
H_{Tmax}	daily maximum heat rejection limit
$H_{\Delta T\uparrow}$	monthly average heat rejection limit
$H_{\Delta T/hour}$	allowable heat rejection rate in million BTU/hour
m	mass of body gaining or losing heat (lb)
MZ	allowed mixing zone percentage divided by 100
Q	flow rate, cfs
Q_{7-10}	seven-day, 10 year low flow (cfs)
Q_e	facility discharge flow in million gallons per day (MGD)
$Q_e + Q_r$ MZ	stream flow downstream of facility discharge in cubic feet per second
Q_r	design stream flow in cubic feet per second (cfs)
ΔT	temperature change
$\Delta T\uparrow$	allowable temperature increase
Te	facility discharge temperature (ºF)
$T_{e\text{-}average}$	average allowable effluent temperature
Tmax	maximum allowable downstream temperature after mixing zone(ºF)
$T_{e\text{-}max}$	maximum allowable effluent temperature
$T_{e\text{-}max1\%}$	allowed effluent temperature for 1% of the hours in 12-months period, °C
$Te\text{-}_{rate}$	maximum allowable effluent rate of temperature change (°F/ hr)

T_{in}	facility intake temperature (°F)
T_r	upstream stream temperature (°F)
T_{II} or T_{III}	Maximum allowed river temperature in the Mississippi River Zone II or III defined in WQS is shown in Table 7.1

6.1. SUMMARY OF TEMPERATURE CRITERIA

The temperature water quality criteria are included in 567 IAC 61.3(3)b(5). The following is a summary of the criteria.

(1) $\Delta T\uparrow$: temperature shall not be increased more than 3°C (5.4 °F) for warm water streams and 2°C (°F) for cold water streams
(2) T_{max}: maximum temperature cap. For all warm waters except the Mississippi River and for cold waters, the temperature caps are 32°C and 20°C respectively. These criteria apply at all times. For the Mississippi River, the temperature cap values vary by month and there are two temperature cap values; one is the absolute temperature cap never to be exceeded, and the other prohibits exceedance for more than 1% of the hours (86.4 hours) in the 12-month period ending with any month.
(3) ΔT/hour: rate of change. The rate of temperature change shall not exceed 1°C/hour.

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

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

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

The following describes how the department will implement the temperature criteria.

6.2 HEAT TRANSFER THEORY AND HEAT-BASED LIMITS

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

$$H = mc\Delta T \qquad (6.1)$$

For simplicity in water quality calculations, the mass (m) of the stream or wastewater is usually expressed as a flow rate (Q) and is expressed in terms of million gallons per day (MGD) or cubic feet per second (cfs). The specific heat (c) of water can be assumed to be 1 since a BTU or British Thermal Unit is defined as the amount of heat required to raise one pound of water by 1°F. Incorporating a flow rate instead of a mass yields results in terms of the rate of heat transfer or heat rejection rate. Equation 6.1 incorporating appropriate conversion factor becomes:

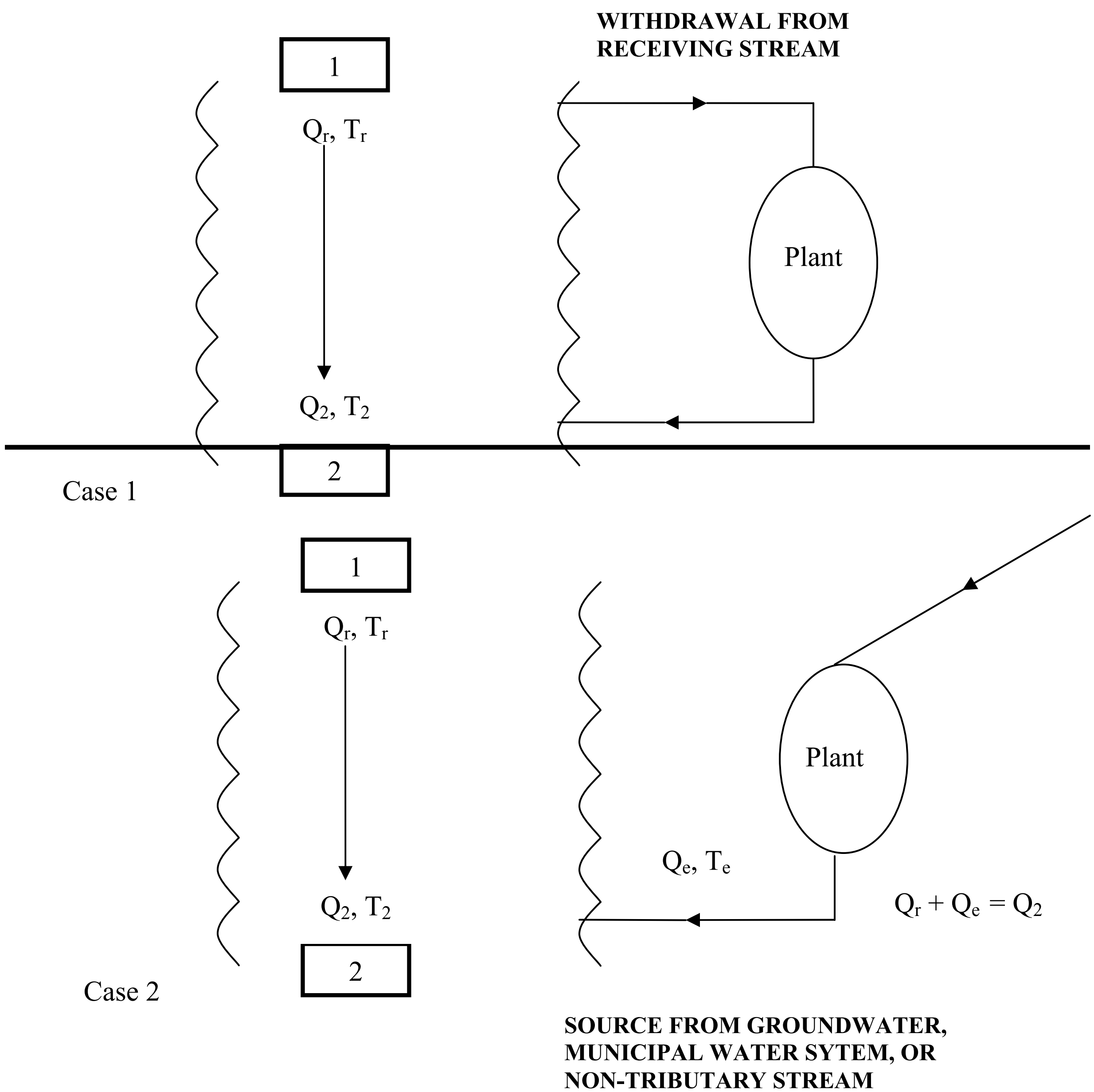
$$H = Q\Delta T \qquad (6.2)$$

Thus, in any closed system, the amount of heat gained or lost may be determined from the heat transfer equation:

$$(Q\Delta T)\ gained = (Q\Delta T)\ lost$$

For the purpose of establishing effluent limits, all thermal discharges fall into one of two categories based upon the source of cooling water, as illustrated by Figure 1. Case 1 situations are those where the source of cooling water is the receiving stream upstream of the point of discharge. Case 2 situations are those where the source of cooling water is not the receiving stream, such as a municipal water system, a well, or from a different waterbody. The form of the heat transfer equation to be used to calculate effluent limits differs for Case 1 and Case 2 situations as explained in the following sections:

FIGURE 1


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

Case 1 - Withdrawal From Receiving Stream

Note! When using the following formulas if million gallons per day (MGD) are used as the units of stream flow instead of cfs, the conversion factor is 8.34 instead of 5.39.

T_{max} criterion implementation:

Figure 1 illustrates the physical layout for a typical Case 1 situation. In the Case 1 situation, effluent limits should be expressed as heat rejection rates, usually in million BTUs/day, unless intake temperature is not available using this equation:

$$H_{Tmax} = Q_r (T_{max} - T_r) * MZ * 5.39 \qquad (6.3)$$

$\Delta T\uparrow$ criterion implementation:

To calculate effluent limits based on the $\Delta T\uparrow$ criterion replace (T_{max} - T_r) in equation 6.3 with the $\Delta T\uparrow$ criterion of 3 ºC (or 5.4 ºF):

$$H_{\Delta T_\uparrow} = Q_r*(5.4\ ºF) *MZ * 5.39 \qquad (6.4)$$

Rate of change calculation: Effluent limits based on the ΔT criterion of 1 ºC/hour (or 1.8 ºF/hour) are calculated using equation 6.5.

$$H_{\Delta T/hour} = Q_r*(1.8\ ºF/hour) *MZ * 5.39/24 \qquad (6.5)$$

A default mixing zone (MZ) of 25% of either the annual or monthly $Q_{7\text{-}10}$ flow for interior streams, and 10% of either the annual or monthly $Q_{7\text{-}10}$ flow for the Mississippi and Missouri Rivers will be used unless site-specific mixing zone study data are available. For Mississippi and Missouri River side channels, sloughs and backwaters 10% of the volume of water that flows through the side channel will be used as the default mixing zone. For interior stream side channels, 25% of the volume of water that flows through the side channel will be used as the default mixing zone.

The default upstream temperature values found in the Default Statewide Water Chemistry Values section of this document will be used unless site-specific upstream temperature data are available.

Determining Compliance:

For all Case 1 situations, the facility discharge flow Q_e, the discharge temperature T_e, and the intake temperature T_{in} may be monitored at least hourly to determine compliance with the allowable heat rejection limit. The discharge heat rejection rate is calculated as follows:

$$H_e = Q_e (T_e - T_{in}) * 8.34 \qquad (6.6)$$

To determine compliance with the rate of change limit ($H_{\Delta T/hour}$), the discharge heat rejection (H_e) needs to be converted from million BTU/day to million BTU/hour by dividing by 24.

The $H_{\Delta T_\uparrow}$ limit will be used as the monthly average permit limit, and the H_{Tmax} limit will be used as the daily maximum limit. The $\Delta T\uparrow$ allowable temperature increase is for the purpose of maintaining a well-rounded population of warmwater 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.

In Case 1 situations, permit limits based on heat rejection rate (million BTU/day) should be established whenever possible because these limits provide a facility more flexibility in achieving compliance. Permit limits expressed as a fixed temperature (ºF) must be calculated using a facility's discharge flow rate whereas permit limits expressed as a heat rejection rate are not. When calculating fixed temperature limits the department must assume that the maximum discharge flow rate can occur when stream flow is at a minimum resulting in permit limits that are more restrictive whenever the discharge flow is less than the flow used in the calculations or when stream temperature or stream flow are different from the values used in the calculations.

Temperature-based limits in Case 1 situations may be considered under the following conditions:

1. Facility discharge flow rates can be accurately and reliably measured;
2. Facility discharge flow rates do not vary substantially within each month, or; the permittee does not object to a permit that specifies limits that may overly restrictive.
3. The facility cannot conduct the monitoring that is required to calculate the heat rejection rate (intake temperature, facility discharge flow, and discharge temperature).

Case 2 - Withdrawal From Source Other Than Receiving Stream

The Case 2 situation is different from Case 1 in that the intake temperature is likely not the same as the receiving stream temperature upstream of the discharge. Figure 1 presents the physical layout for a typical Case 2 situation.

For case 2 situations, the temperature of the intake water is not known, the heat rejection for the effluent cannot be calculated. In these cases, permit limits must be expressed based on the temperature of the discharge water, and compliance monitoring is accomplished by measuring the temperature of the discharge water. The limitations of this method are as listed in the previous section, in that the adequacy of the permit limits depends on an accurate facility flow balance, and the analyst must assume conservative or maximum facility flow rates. The derivation of temperature limits is shown in Section 6.3.

6.3. TEMPERATURE-BASED LIMITS CALCULATIONS

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

$$T_{max}: \quad (Q_e + Q_r\, MZ) * T_{max} = Q_r * MZ * T_r + Q_e * T_e \qquad (6.7)$$

$$\Delta T\uparrow: \quad T_e = 3^{o}C * D + T_r \qquad (6.8)$$

Warm Water Interior Streams and the Big Sioux River: Monthly average, daily maximum and rate of temperature change effluent limits will be calculated based on IAC Chapter 61.3(3)"b"(5) temperature criteria. The statewide average background temperatures for streams designated as warm waters described in the Default Statewide Water Chemistry section of this document are used in the calculations.

Monthly Average Limits: WQS requires 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 regulatory mixing zone. The default mixing zone is 25% of the Q_{7-10} flow in the receiving stream and will be used in the calculations unless site-specific mixing zone data are available. See Section 13 for definition of AWW effluent flow to be used in these calculations. The calculation is described by equation 6.9:

$$T_{e\text{-}average} = T_r + 3°C * D \qquad (6.9)$$

Daily Maximum Limits: WQS requires that "in no case shall heat be added in excess of that amount that would raise the stream temperature above 32 °C". The same mixing zone and AWW effluent flow will be used to calculate monthly average limits and daily maximum limits. The calculation is described by equation 6.10:

$$T_{e\text{-}max} = T_r + (32\ °C - T_r) * D \qquad (6.10)$$

Rate of Change (ΔT) Limits: WQS requires that "the rate of temperature change shall not exceed 1°C per hour". The same mixing zone and effluent flow will be used to calculate the monthly average limits and daily maximum limits will be used to calculate *ΔT limits*. The calculation is described by Equation (6.11):

$$T_{e\text{-}rate}\ (°C/\ hr) = 1\ (°C/hr) * D \qquad (6.11)$$

Equation (6.11) can be used for large heat sources when there are continuous temperature monitoring data. For smaller heat sources, the following special conditions may be included in the wasteload allocations to implement the rate of change criteria.

Special conditions: 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 1ºC per hour.

Cold Water Streams: 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:

(1) Background temperature values: The statewide background temperature values derived for cold water streams described in the Default Statewide Water Chemistry section of this document are used.
(2) Criteria: IAC Chapter 61.3(3)"b"(5)"2" states that "No heat shall be added to streams designated as cold water fisheries that would cause an increase of more than 2°C. The rate of temperature change shall not exceed 1°C per hour. 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 in the formulas.

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 mixing zone. For the Missouri River, the default mixing zone is 10% of the Q_{7-10} flow instead of 25% and this default flow will be used unless site-specific mixing zone data are available.

The Mississippi River: Monthly average, daily maximum and rate of change limits will be calculated according to the temperature criteria described in IAC 61.3(3)"b"(5)"5". An additional criterion for the Mississippi River is that the water temperature shall not exceed the maximum limits in Table 6.1 during more than 1 percent of the hours in the 12-month period ending with any month. The average background temperatures for the Mississippi River Zone II (from Iowa north border to Wisconsin – Illinois border) and Zone III shown in the Default Statewide Water Chemistry section of this document are used in the calculations.

Monthly Average Limits: WQS requires 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 mixing zone is 10% of the Q_{7-10} flow for the Mississippi River at the discharge location unless site-specific mixing zone data are available. . See Section 13 for definition of AWW effluent flow to be used in these calculations The calculation is described by equation 6.12:

$$T_e = T_r + (3°C) * D \qquad (6.12)$$

Daily Maximum Limits: WQS requires that at no time shall the water temperature exceed the maximum limits in table 7.1 by more than 2°C. The same mixing zone and AWW effluent flow will be used to calculate the monthly average limits and daily maximum limits. The calculation is described by equation 6.13

$$T_{e\text{-}max} = Tr + [2 + (T_{II} \text{ or } T_{III}) - Tr] *D \qquad (6.13)$$

Table 6.1: Maximum Allowed River Temperature 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)

Month	Zone II	Zone III
	Temperature *(°C)	Temperature *(°C)
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

Rate of Change ($\Delta T\uparrow$) Limits: WQS requires that "the rate of temperature change shall not exceed 1°C per hour". The same mixing zone and effluent flow used to calculate the monthly average limits and daily maximum limits will be used to calculate *ΔT limits*. The calculation is described by equation 6.14:

$$T_{e\text{-}rate}\ (°C/\ hr) = 1\ (°\ C/\ hr) * D \qquad (6.14)$$

Equation (6.14) can be used for large heat sources when there are continuous temperature monitoring data. For smaller heat sources, the following special conditions may be included in the wasteload allocations to implement the rate of change criteria.

Special conditions: 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 1ºC per hour.

The water temperature in the Mississippi River in Zone II shall not exceed the maximum limits shown in Table 6.1 during more than 1 percent of the hours in the 12-month period ending with any month: The same mixing zone and AWW effluent flow will be used to calculate both $T_{e\text{-}max}$ and $T_{e\text{-}max1\%}$. The calculation is described by equation 6.15:

$$T_{e\text{-}max1\%} = Tr + (T_{II}\text{ or }T_{III}\text{-}Tr) * D \qquad (6.15)$$

The effluent temperature limit based on meeting the $\Delta T\uparrow$ criterion of 3ºC (5.4ºF) will be used as the monthly average limit and the T_{max} limit will be used as the daily maximum limit

6.4. **IMPLEMENTATION OF 567 IAC 61.2(5)"A" AND IAC 61.2(5)"B"**

According to 567 IAC61.2(5)"a" and IAC61.2(5)"b", during the period November 1 to March 31, for the purpose of applying the 3°C temperature increase criterion, the minimum protected receiving stream flow rate below such discharges may be increased to not more than three times the rate of flow of the discharge, where there is reasonable assurance that the discharge is of such constant temperature and flow rate and continuous duration as to not constitute a threat of heat cessation and 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 that the discharge is of such constant temperature and flow rate and continuous duration, when the receiving stream flow is less than 2x the discharge flow a stream flow of two times the discharge flow rate in lieu of the $Q_{7\text{-}10}$ will be 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 Missouri Rivers.

6.5. 316(a) DEMONSTRATIONS

Section 316(a) of the Federal Water Pollution Control Act provides 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 *Implementation Guidance Evaluation and Process Thermal Discharge, (316(a)) Federal Clean Water Act, USEPA, 1974* and must contact the department for approval prior to undertaking any studies.

7. Determination of Stream Flow Velocity

Stream flow velocities are often needed for performing bacteria or total residual chlorine decay calculations. Site-specific velocity is always preferred. Sometimes, the stream velocity can be 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 gravity discharge pipe or storm sewer
c. 1 – 2 fps for pressured pipe flows, such as pressured sewer outfall pipe.

8. E. coli standard Implementation

a. Continuous Discharges

IAC 61.3(3)"a"(1) shows the E coli criteria Table that are applicable to designated Class "A" waters. Waters which are designated as Class "A1," "A2," or "A3" in subrule 61.3(5) are to be protected for primary contact, secondary contact, and children's recreational uses. The general criteria of subrule 61.3(2) and the following specific criteria apply to all Class "A" waters. (1) The Escherichia coli (E. coli) content shall not exceed the levels noted in the Bacteria Criteria Table when the Class "A1," "A2," or "A3" uses can reasonably be expected to occur.

After Iowa adopted the current E. coli criteria in 2003, EPA published the BEACH Act Rule in 2004 (69 FR 67217, November 16, 2004). In the BEACH rule, EPA indicated that it expected that the single sample maximum values would be used for making beach notification and closure decisions. EPA recognized, however, that States and Territories also use criteria in their water quality standards for other purposes under the Clean Water Act in order to protect and improve water quality. Other than in the beach notification and closure decision context, the geometric mean is the more relevant value for ensuring that appropriate actions are taken to protect and improve water quality. The geometric mean is generally more relevant because it is usually a more reliable measure of long term water quality, being less subject to random variation, and more directly linked to the underlying studies upon which the 1986 bacteria criteria were based.

Accordingly, Iowa revised Chapter 62 regarding the implementation of the sample maximum criterion, which became effective on October 14, 2009. The revised Chapter 62 states "…that the daily sample maximum criteria for E. coli set forth in Part E of the 'Supporting Document for Iowa Water Quality Management Plans' shall not be used as an end-of-pipe permit limitation."

Therefore, based on the BEACH Rule regulation and the revision of Chapter 62, only the geometric mean limit of 126 org./100 ml applies to continuous discharges.

b. Intermittent Discharges

The use of the geometric mean limit of 126 org./100 mL makes establishing limits somewhat difficult for intermittent discharges such as controlled discharge lagoons and CSOs, since there may not be adequate sample data to calculate a geometric mean due to their limited duration of discharge (current rules require no less than five samples evenly distributed during a 30-day period to calculate the geometric mean). Thus, a single sample maximum value based on the same risk levels or same level of protection as the geometric mean value is used for these types of discharges.

The current *E. coli* geometric mean criterion of 126 org/100 mL was developed based on a risk level of 8 illnesses per 1000 people. In order to derive an equivalent sample maximum value corresponding to the same risk level as the geometric mean of 126 org/100 mL, the variability of *E. coli* levels should be taken into account. Based on the EPA 1986 Bacteria Document, when a site-specific coefficient of variance is not available, a default log coefficient of variance of 0.4 can be used for freshwaters. Based on the default coefficient of variance of 0.4 and the geometric mean value of 126 org/100 mL, the sample maximum values at different confidence levels can be calculated as shown in Table 8.1. The 99th percentile value of 1,073 org/100 mL is selected as the sample maximum value since it is consistent with Iowa's current permit derivation procedure. Thus, an effluent sample maximum value below 1,073 org/100 mL will ensure that there is a greater than 99% probability that the waterbody will meet a geomean criterion of 126 org./100 mL. Correspondingly, an effluent sample maximum value of 5,365 org./100 mL is used to meet Class A2 uses.

Table 8.1. Water Quality Criteria for Bacteria for Fresh Recreational Waters

Indicator	Geometric Mean Density	Single Sample Maximum Allowable Density*			
		75th percentile	90th percentile	95th percentile	99th percentile
E coli	126	235	410	576	1,073

* Sample maximum values are calculated based on a log coefficient of variance of 0.4

9. E coli Decay Rate

The current E. coli decay rate was derived from the EPA 1985 document titled "Rates, constants and kinetics formulations in surface water quality modeling. 2nd ed. EPA/600/3-85/040". The document summarized about twelve decay rates for streams and rivers, and six decay rates for lakes and ponds. The decay rates came from studies conducted from 1920's to 1980's. Based on the author's data analysis on the original decay rates data, Iowa's current decay rate of 5.28/day was about the 90th percentile value of all the decay rates.

IDNR conducted a re-analysis on the decay rate data published in the EPA 1985 document and focused on stream and river decay rates only.

The Analysis of the Original Decay Rates:

The mean and median decay rates based on the original decay rates are shown in Table 9.1:

Table 9.1. Summary Statistics of the Original Decay Rates

k(1/day)	Rivers & Streams	Lakes & Ponds
Mean	3.37	2.31
Median	1.03	1.46

The Analysis of the Log-transferred Decay Rates:

The decay rates were first log-transferred since the decay rates are log normally distributed. The decay rates were re-analyzed using the log-transformed data. The purpose of using log-transferred data is to calculate the confidence interval values for the median values.

All of the bacteria decay study data in the EPA document are in-situ studies (that is, they are field study data, not laboratory data), and they are decay rates for coliforms. Table 9.2 shows the summary statistics of the original data assuming a log-normal distribution for the decay rates:

Table 9.2. Summary Statistics Using Log-transferred Decay Rates

k(1/day)	Rivers & Streams	Lakes & Ponds
Mean	2.90	2.57
Median	1.30	1.15
95% Confidence Interval	0.85 – 1.97	0.62 – 2.15

The statistical analysis of the log transformed data for streams and rivers show the median value is about 1.30/day, and the 95% confidence intervals for the median value are 0.85/day and 1.97/day. The statistical analysis of the log transformed data for lakes and ponds show the median value is about 1.15/day, and the 95% confidence intervals for the median value are 0.62/day and 2.15/day. The recalculated median values using log-normal distribution are a little different than the median values calculated using original data. To minimize the data transfer errors, the median values based on the original data shown in Table 9.1 are used as the bacteria decay rates for streams and ponds. The decay rates for streams and ponds are1.03/day and 1.46/day, respectively.

10. Flow Variable Limits

Wasteload allocations based on flow variable limits as (lb/day/cfs) will not be used for new NPDES permits or renewal of NPDES permits. The implementation of such limits poses problems both for the department and for the permittee. First, a facility must have a means to accurately determine stream flow at the point of discharge on a daily basis. This generally means that there must be an existing stream gaging station in close proximity to the discharge or a gaging station must be installed and maintained by the permittee.

Second, to comply with flow variable limits a facility must be designed with the flexibility to vary its operations as stream flow increases or decreases. For example, a facility with a flow variable permit limit for ammonia nitrogen must be capable of varying operations to provide more or less treatment of ammonia as stream flow changes or must be able to store all or a portion of its wastewater when stream flow is low.

Third, a facility must be designed and capable of meeting permit limits when stream flow is at the critical low flow (i.e. $Q_{30\text{-}10}$, $Q_{7\text{-}10}$, $Q_{1\text{-}10}$) even though stream flow may not require operating at this level most of the time. As mentioned earlier, Iowa's numeric water quality criteria must be met whenever flow in the receiving stream is equal to or exceeds a designated low flow.

Fourth, except for facilities with existing permits that contain flow variable limits, the implementation of flow variable limits is inconsistent with Iowa's proposed antidegradation policy because it would allow a facility to increase the concentration and amount of a pollutant in the discharge without demonstrating the necessity for such limits.

Wasteload allocations will continue to be calculated using stepwise flow limits 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. However, the facility must clearly demonstrate that there is sufficient storage available to operate in this manner and must have an accurate means of determining stream flow at the point of discharge.

11. The Use of Monthly Critical Low Flows for Ammonia Nitrogen Limits and Temperature

Chapter 61.2(5) defines the critical low flows used for wasteload allocation purposes. Usually annual critical low flows are used since the numerical criteria for most pollutants do not vary from month to month. One exception is the ammonia nitrogen criteria and the temperature criteria (both stream background temperature and temperature criteria could change from month to month). The ammonia nitrogen criteria are pH and/or temperature dependent. Due to low air temperature, Spring and Winter seasons are usually the time period when biological treatment processes have the most difficulty in removing ammonia nitrogen. The Spring season is most likely to have elevated stream flow available for dilution. The use of monthly critical low flows is consistent with the allowed risk levels afforded in the EPA TSD guidance (EPA TSD, 1991). The issue is when the monthly critical low flows are applied when developing WLAs. There are three options: (1) using monthly critical low flows for all facilities for ammonia nitrogen and temperature; (2) continue to use annual critical low flows;

(3) only apply monthly critical low flows on a case by case basis (considering the staff efforts and no significant difference in limits). For now, the Department is recommending the following option:

Monthly critical low flows are consistently used for temperature due to the fact that the heat loss in the mixing zone is not taken into account in the current WLA procedure. For ammonia nitrogen WLA, the Department will explore the use of monthly critical low flows on a case by case basis and will consider the factors such as:

(1) It is the receiving stream a perennial stream;
(2) Is the receiving stream an effluent dominated stream;
(3) Is there a nearby USGS gage has adequate flow record to be used reasonably to estimate the monthly low flows at the discharge location.

It is important to note that in many cases, the monthly critical low flows are also zero especially in effluent dominated streams. The use of monthly critical low flows may make a difference in ammonia nitrogen limits when the receiving streams are perennial. In addition, if the use of monthly critical low flow results in less stringent limits, antidegradation Tier 2 review is required to demonstrate that the degradation is necessary and important. In these cases, the current limits will govern.

Monthly stream critical low flows will be applied only for ammonia and temperature water quality based limits due to the fact that the ammonia and temperature criteria vary from month to month.

12. Define the discharge flows for industrial discharges used in WLAs

Wasteload allocation analyses are performed for monthly conditions using projected 20 year Average Dry Weather (ADW) and Average Wet Weather (AWW) wastewater discharge flows entering a receiving stream which is at the design low stream flow. However, industrial discharges such as cooling water discharges do not usually have wastewater design flows since no wastewater treatment design is needed. For industrial discharges, maximum monthly average flow and daily maximum flow are usually provided in the NPDES permit application. The only guidance found is the EPA TSD document (EPA, 1991). It recommends the use of maximum discharge flows. In order to be consistent with the flows used for municipal discharges, a maximum monthly average discharge flow and a maximum daily discharge flow are used in the WLA calculations for industrial discharges where wastewater design flows are not applicable.

13. Mixing Zone and Zone of Initial Dilution Policies for Reservoirs

IAC Chapter 61.2(4) prohibits mixing zone and zone of initial dilution for waters designated as lakes or wetlands. For reservoirs (Coralville Lake, Lake Red Rock and Saylorville Lake) on streams designated as Class B waters and big pool conditions such as on the Mississippi River, the default mixing zone and zone of initial dilution will use 1% and 0.1% of the Q_{7-10} and Q_{1-10} of the stream flow for toxics; 1% and 0.1% of the Q_{30-10} and Q_{1-10} of the stream flow for ammonia nitrogen unless site-specific mixing zone data are available. Site specific mixing zone data through modeling or field study may be used in lieu of default mixing zone sizes.

14. Site-Specific Data Collection

The permittee may choose to collect site-specific ambient and/or effluent water chemistry data in lieu of the statewide default values and submit the data for Department consideration. Wastewater treatment facilities are encouraged to plan ahead when considering any site-specific data gathering effort. Some of these efforts require seasonal data particularly collected during low stream flow conditions. A detailed sampling plan should be submitted to IDNR for approval prior to beginning any site-specific data collection. Both local and regional data may be collected and submitted to the Department.

a. *Local Values:* If the applicant desires that local values be used, they must supply a minimum of 2 years of water chemistry readings (hardness, sulfate and chloride) and sample at least once per month.

> 1) Background Water Chemistry: The applicant must submit a minimum of 2 years of grab samples for ambient background water chemistry and the samples must be collected at least once per month. More frequent monitoring in a shorter time period than 2 years may be allowed if the applicant can demonstrate that the monitoring data are collected in a representative year. The factors IDNR will consider to determine if the data are collected in a representative year include: stream flow (no drought or flooding conditions), weather patterns of the year, etc). Monitoring values should be obtained from upstream of the outfall at a representative location of the true upstream background condition of the discharge.
>
> In cases for certain pollutants, seasonal water chemistry data may be required to catch the most critical conditions such as low stream flow and high temperature conditions. The sample plan must be able to catch the required critical.
>
> 2) Effluent Water Chemistry: For effluent water chemistry determinations, a 24-hour composite sample of the final effluent is required. For intermittent discharges, a 24-hour composite sampling may not be feasible; a representative grab sample is also acceptable.

b. *Regional Background Values:* Regional water chemistry data could be available that represents the upstream background conditions. For example:

(1) Another facility, at a reasonable distance upstream of the facility of interest, has collected background readings of water chemistry data (such as hardness, sulfate and chloride) that is representative of the background chemistry for the facility of interest;

(2) Ambient monitoring data are available within the same watershed (such as STORET data) that is representative of the upstream background conditions of the facility of interest;

The factors that could influence if the regional background data are acceptable to use include: the distance to the outfall, another discharge between the regional station and the oufall, another tributary, which could influence the water chemistry at the outfall. If the site specific ambient data are judged to be acceptable, the site specific data will be used instead of the statewide default water chemistry values.

15. Default Statewide Water Chemistry Values

(1) Statewide Ambient pH and Temperature

Statewide ambient average pH and temperature for interior warm water streams are derived using the DNR ambient monitoring station data from 2000 – 2009.

New average temperature data for the Mississippi River and the Missouri River are derived using the temperature monitoring data from USGS and the Corps of Engineers monitoring stations. Average temperature and pH values for lakes are calculated using Iowa State University Lake Study (the detail calculations are described in Estimation *of Background Temperature and pH Values for Rivers/Streams in Iowa*). The new ambient cold water average temperature data are derived from source data collected by UHL from 138 sites located on streams designated as cold water, from June of 1999 to October 2008.   Monthly averages were calculated based on these data.

Table 15.1: Statewide Average Background Temperature for Streams Designated As Warm Water

Month	Statewide Warm Water Background Temperature (°C)
Jan.	0.7
Feb.	0.5
Mar.	2.3
Apr.	9.3
May	15.3
Jun.	19.5
Jul.	23.8
Aug.	24.7
Sep.	20.8
Oct.	14.6
Nov.	7.7
Dec.	1.8

Table15.2: Statewide Average Background Temperature for Cold Water Designated Streams

Month	Statewide Cold Water Background Temperature (°C)
Jan.	5.5
Feb.	4.4
Mar.	6.4
Apr.	9.5
May	13.3
Jun.	16.8
Jul.	18.1
Aug.	17.5
Sep.	15.1
Oct.	11.0
Nov.	7.8
Dec.	5.3

Table 15.3: Average Background Temperature for the Missouri River along Iowa West Border& Lakes

Month	Statewide Missouri Background Temperature (°C)	Statewide Lakes Background Temperature (°C)
Jan.	1.0	--
Feb.	1.3	--
Mar.	4.8	--
Apr.	10.8	13.6
May	17.3	18.5
Jun.	22.7	23.0
Jul.	26.1	26.7
Aug.	25.6	26.3
Sep.	20.9	21.1
Oct.	14.1	16.3
Nov.	6.9	--
Dec.	1.5	--

Table 15.4: Average Background Temperature for the Mississippi River Zone II

Month	Statewide Warm Water Background Temperature (°C)
Jan.	0.5
Feb.	0.7
Mar.	3.1
Apr.	9.9
May	16.1
Jun.	22.2
Jul.	25.1
Aug.	24.3
Sep.	20.3
Oct.	12.5
Nov.	5.7
Dec.	1.2

Table 15.5: Average Background Temperature for the Mississippi River Zone III

Month	Statewide Warm Water Background Temperature (°C)
Jan.	0.4
Feb.	1.0
Mar.	3.8
Apr.	11.1
May	16.7
Jun.	22.5
Jul.	25.4
Aug.	25.3
Sep.	21.9
Oct.	13.8
Nov.	7.1
Dec.	1.2

Table 15.6: Statewide Average Background pH Values

Month	Default Ambient pH Values for Warmwater	Default Ambient pH Values for Coldwater	Default Ambient pH Values for Lakes
Jan.	7.6	8.1	--
Feb.	7.9	8.1	--
Mar.	7.9	7.9	--
Apr.	8.1	8.1	--
May	8.1	8.1	8.3
Jun.	8.0	7.8	8.3
Jul.	8.1	8.0	8.3
Aug.	8.2	8.0	8.2
Sep.	8.2	8.0	8.6
Oct.	8.2	8.1	--
Nov.	8.2	8.1	--
Dec.	8.1	8.2	--

(2) Statewide Effluent pH and Temperature

Statewide effluent pH and temperature values were developed several years ago for aerated lagoon, mechanical treatment plants, and industrial discharges.

The statewide effluent pH and temperature values for covered lagoons were developed in 2003 based on the data submitted from City of Wheatland consultant. The consultant submitted historic effluent pH data for covered aerated lagoon facilities in several other states since no floating covered lagoons existed in the state of Iowa at that time. The facilities include Brownsville, Wisconsin; Iron Ridge, Wisconsin; Poplar, Wisconsin, Camila, Georgia; Fishing River, Missouri and Gibsland, Louisiana. The site specific effluent pH values for floating covered aerated lagoons were extrapolated from the monitoring data (2 years) for the above facilities.

Table 15.7: Statewide Effluent pH and Temperature Values for Different Treatment Plant Discharges

Months	Aerated Lagoon		Mechanical Plant		Industrial Discharge		Covered Lagoon	
	pH	Temperature	pH	Temperature	pH	Temperature	pH	Temperature
Jan.	7.5	4.5	7.67	12.4	7.9	17.83	7.5	9.6
Feb.	8	8.1	7.71	11.3	8.1	17.83	8	10.3
Mar.	8.4	8.7	7.69	13.1	8	27.67	8	11.2
Apr.	8.3	14.6	7.65	16.2	8.2	33.89	8	10.8
May	8.5	18.8	7.67	19.3	8.3	35.89	8	18.3
Jun.	8.5	22.8	7.7	22.1	8.2	38.67	8	18.5
Jul.	8.5	25.3	7.58	24.1	8.2	40.61	8	19.4
Aug.	8.6	25.3	7.63	24.4	8.2	39.61	8	19.2
Sep.	8.6	22.2	7.62	22.8	8.3	34.5	8	19.3
Oct.	8.6	16.6	7.65	20.2	8.2	31.89	8	12.4
Nov.	8.6	12.4	7.69	17.1	8.2	29.39	8	11.7
Dec.	8.4	8.4	7.64	14.1	8.1	24.67	8	10.8

(3) Statewide Background Chemical Concentrations

Iowa Water Quality Standards have defined numerical criteria for 89 priority pollutants. To properly implement these criteria and calculate NPDES permit limits for each wastewater discharge, background concentrations of the pollutants in Iowa surface waters have to be established. Two main sources of monitoring data are available. One is the Iowa's STORET data, the other is the USGS water quality monitoring data. A brief description of the data from the two sources is as follows:

STORET network data
The STORET network has 90 monitoring sites for Iowa interior streams. It has been collecting data since 1999. Sixty five of the 89 priority pollutants that have numerical criteria are among the parameters that the network monitors.

However, STORET dataset has a large number of observations that are reported as non-detect (ND). Of the 66 parameters of our interest, 48 parameters do not have a single monitoring reading above detection limits; 9 parameters have more than 95% of the readings that are below detection limits; 5 parameters have some but less than 95% of readings below detection limits; only 4 parameters do not have readings below detection limits. Large number of ND in a dataset always brings uncertainty.

USGS monitoring network data
USGS has been monitoring water quality parameters in Iowa waters. Comparing with STORET data, it has more reported values. However, the datasets do not explicitly report detect limits.

Data used for statewide chemical background levels
Data from both the STORET Network and the USGS monitoring network will be used. Datasets for each pollutant will be reviewed for their detection frequency, number of above detection limit readings.

Statistical Analysis Methods
Most statistical methods used to handle dataset needs to have a portion of the data as reported values. For a dataset that has 100% ND or a large percentage of ND, none of the statistical methods would be applicable or would produce satisfactory results. EPA suggests that in cases when the detection frequency is low (e.g., < 4%-5%) and the number of detected observations is low (<4-6 readings), the project team and the decision makers together should make a decision on a site-specific basis.

After reviewing the analysis results and considering the EPA guidance, it is decided that datasets having more than 95% of the ND will not be analyzed by any statistic method. A zero background level will be assigned to these chemicals.

Datasets with more than 5% of the detected value will be analyzed by the Kaplan-Meier method. As one of the nonparametric procedures, Kaplan-Meier method is widely recommended for analysis of water quality data for the following reasons:

- Most water quality data does not follow a certain distribution
- Kaplan-Meier methods have been proven to produce best estimates of the upper confidence levels
- Kaplan-Meier method produces better results for smaller sample size or highly skewed data.

Table 15.8 lists the estimated statewide background concentrations for different priority pollutants.

Chemical	Median (ug/l)	Chemical	Median (ug/l)
1,1,1-Trichloroethane	0.00	Endosulfan	0.00
1,1,2-Trichloroethane	0.00	Endothall	--
1,1-Dichloroethylene	0.00	Endrin	0.00
1,2,4-Trichlorobenzene	0.00	Ethylbenzene	0.00
1,2-Dichloroethane	0.00	Ethylene dibromide	--
1,2-Dichloropropane	0.00	Fluoride	200
2,3,7,8-TCDD (Dioxin)	--	gamma-Hexachlorocyclohexane (Lindane)	0.000
2,4,5-TP (Silvex)	--	Glyphosate	0.03
2,4-D	0.000	Heptachlor	0.00
3,3-Dichlorobenzidine	0.00	Heptachlor epoxide	0.00
4,4' DDT	--	Hexachlorobenzene	0.00
Alachlor	0.000	Hexachlorocyclopentadiene	0.00
Aldrin	0.000	Lead	3.000
Aluminum	700	Mercury (II)	0.000

Antimony	0.00	Methoxychlor	0.00
Arsenic (III)	3.000	Nickel	3.000
Asbestos	--	Nitrate as N	1,800
Atrazine	0.098	Nitrate+Nitrite as N	5,300
Barium	100.0	Nitrite as N	20
Benzene	0.000	o-Dichlorobenzene	0.00
Benzo(a)Pyrene	0.00	Oxamyl (Vydate)	0.00
Beryllium	0.00	para-Dichlorobenzene	--
Bromoform	--	Parathion	0.00
Cadmium	0.00	Pentachlorophenol (PCP)	0.00
Carbofuran	0.00	Phenols	0.00
Carbon Tetrachloride	--	Picloram	0.00
Chlordane	0.00	Polychlorinated Biphenyls (PCBs)	--
Chlorobenzene	0.00	Polynuclear Aromatic Hydrocarbons (PAHs)	--
Chlorodibromomethane	0.00	Selenium (VI)	1.000
Chloroform	190	Silver	0.000
Chloropyrifos	--	Simazine	0.000
Chromium (VI)	0.00	Styrene	0.000
cis-1,2-Dichloroethylene	0.00	Tetrachloroethlyene	0.000
Copper	10.00	Thallium	0.000
Cyanide	0.000	Toluene	0.030
Dalapon	--	Total Residual Chlorine (TRC)	--
Di(2-ethylhexyl)adipate	--	Toxaphene	0.000
Bis(2-ethylhexyl)phthalate	0.00	trans-1,2-Dichloroethylene	0.000
Dibromochloropropane	--	Trichloroethylene (TCE)	0.000
Dichlorobromomethane	0.00	Trihalomethanes (total)	0.000
Dichloromethane	0.00	Vinyl Chloride	0.000
Dieldrin	0.00	Xylenes (Total)	0.000
Dinoseb	0.00	Zinc	0.000
Diquat	--		

(4) Statewide Background Levels for Hardness, Chloride and Sulfate Criteria

Chloride and sulfate toxicity are both heavily dependent on water hardness. To a lesser degree chloride toxicity is also dependent on the sulfate concentration of the waters, while sulfate is dependent on the chloride concentration of the waters. For those situations where site-specific water chemistry may not be available, statewide default water chemistry values were developed. 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

In order to develop statewide background water chemistry values, the IDNR ambient monitoring data from 2000-2007 were analyzed. The data analysis shows the 10th percentile for hardness is 200 mg/L as $CaCO_3$. The regression analyses between hardness and sulfate, and hardness and chloride show that there is a general positive relationship between hardness and sulfate, as well as hardness and chloride. However, the data are fairly scattered. The $90^{\%}$ Prediction Interval is quite large. Thus, if a conservative sulfate (and chloride) concentration corresponding to the hardness value of 200 mg/L as $CaCO_3$ is chosen, the upper limits of the $90^{\%}$ prediction interval could be used as statewide background values. In this case, the sulfate concentration is 63 mg/L, and the chloride concentration is 34 mg/L. Utilizing the above values, the water quality criteria for chloride are 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.

The default hardness value of 200 mg/L as $CaCO_3$ is also used to develop hardness dependent metal criteria and wasteload allocations unless site-specific hardness data are provided.

16. Water Quality Modeling

The ability of a stream to maintain an acceptable dissolved oxygen (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 reaeration from the atmosphere will replace any DO lost through oxidation processes. If, however, the rate of oxygen use exceeds the rate of reaeration, the DO concentration may decrease below minimum allowable standards.

Water quality models are useful tools that can be used to predict the effects of point and nonpoint sources on dissolved oxygen levels in a waterbody. Water quality modeling is an attempt to relate specific water quality conditions to natural processes using mathematical relationships. A water quality model usually consists of a set of mathematical expressions relating one or more water quality parameters to one or more natural processes. Water quality models are most often used to predict how changes in a specific process or processes will change a specific water quality parameter or parameters.

Water quality models vary in complexity from simple relationships which attempt modeling a few processes under specific conditions to very complex relationships which attempt to model many processes under a wide range of conditions. The simpler models are usually much easier to use and require only limited information about the system being modeled but 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 may relate many natural processes to several water quality parameters on a time-dependent basis. These models are usually harder to apply andrequire extensive information about the system being modeled, but also have a broader range of applicability. Dynamic models fall into this category.

To predict the variation in DO, as well as ammonia concentration in streams, a simplified Excel spreadsheet implementing the modified Streeter-Phelps DO Sag equation and a more complex mathematical model such as QUALIIK have been used in Iowa. Input data for the models isdeveloped from existing technical information and site specific field investigations of selected streams. When sufficient data is not available, conservative assumptions are applied until site specific information becomes available.

THEORY AND METHODOLOGY

Modeling Theory

Dissolved oxygen (DO) concentrations in streams are controlled by many factors including atmospheric reaeration, biochemical oxygen demands (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. As a result of this difficulty, 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 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 concentration. Benthal oxygen demands result from anaerobic decomposition of settled organic material at the bottom of the stream. These reactions release carbonaceous and nitrogenous organic materials that create biochemical oxygen demands. The inclusion of benthal oxygen demands in the QUAL-IIK model requires extensive field surveys to determine the real extent of sludge deposits within a stream and coefficients that describe the release into the water. In most instances no data is available to accurately describe sludge deposition areas. Benthal oxygen demands are usually not included in the Excel spreadsheet model. QUAL-IIK has the sediment oxygen demand (SOD) component. The sediment-water fluxes of dissolved oxygen and nutrients are simulated internally rather than prescribed. That is, SOD 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 SOD simulation is bested used when sufficient field data are available to calibrate and verify the rate constants. If field data are not available, default rate constant values can be used.

Nitrogenous BOD is due to the oxidation of ammonia 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 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 (expressed as nitrogen) to nitrate. This theoretical value may conservatively over estimate 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 (NH_3-N). The spreadsheet implementing the Streeter-Phelps equation uses 4.33 as the ratio of nitrogenous BOD to NH_3-N. Assuming secondary wastewater treatment plant effluents contain NH_3-N levels of 10 mg/L during summer operations and 15 mg/L during winter periods, the equivalent nitrogenous BOD (should all the ammonia be converted to nitrates) is approximately 40-46 mg/L (summer) and 62-68 mg/L (winter).

Modified Streeter-Phelps DO Sag Model

1. Dissolved Oxygen Deficit Equation

The spreadsheet uses modified Streeter-Phelps equation to predict DO deficit within the stream. This approach recognizes carbonaceous and nitrogenous BOD, atmospheric reaeration, and initial DO deficit. The effects of photosynthesis and benthal oxygen demands are usually not specifically considered unless site specific data are available. The Streeter-Phelps equation that is implemented in the spreadsheet is as follows:

$$D(t)=\frac{K_1L_o}{K_2-K_1}(e^{-K_1(t)}-e^{-K_2(t)})+\frac{K_NN_o}{K_2-K_n}(e^{-K_n(t-t_0))}-e^{-K_2(t-t_0))})+D_oe^{-K_2(t)}+\frac{(R-P)}{K_2}(1-e^{-K_2(t)})+\frac{SOD}{K_2H}(1-e^{-k_2t})$$

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_1 = Carbonaceous deoxygenation rate constant, base e, day $^{-1}$

K_N = Nitrogenous deoxygenation rate constant, base e, day $^{-1}$

K_2 = Reaeration 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; when a wastewater contains both carbonaceous and nitrogenous oxygen demand, there is usually 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 less than t_0 the nitrogenous term does not enter into the calculation of D(t).

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

$$N_0 = 4.33N_{n0}$$

Where:

N_{no}, initial ammonia nitrogen concentration, mg/L

In this equation, 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. The other traditional Streeter-Phelps components (Streeter, 1925) remain unchanged.

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

$$L(t) = L_o e^{-K_1(t)}$$

$$N(t) = N_o e^{-K_N(t)}$$

where:

L(t) = Ultimate carbonaceous BOD at time, t, mg/L

N(t) = Ultimate nitrogenous BOD at time, t, mg/L

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

Nitrifying bacteria are generally present in relatively small numbers in untreated wastewaters. 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, may be reduced or eliminated in a stream due to a number of reasons including the following: 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 can take place with a resultant build-up of nitrifying organisms. These nitrifying bacteria can immediately begin to oxidize the ammonia present and exert a significant oxygen demand in a stream below the outfall.

It is known that the biological nitrification process is generally more sensitive to environmental conditions than carbonaceous decomposition. The optimal temperature range for growth and reproduction of nitrifying bacteria is 26° to 30° C. It is generally concluded that the nitrogenous BOD will assume greatest importance in small streams which receive relatively large volumes of secondary wastewater effluents during the low flow, warm weather periods of the year (August and September). These conditions were used for the low flow determination of allowable effluent characteristics during summer periods. 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 was observed.

2. Rate Constant Determination

a. Deoxygenation Rate Constants

The carbonaceous deoxygenation rate constant (K_1) for most streams will vary from 0.1 to 0.5 per day (base e, 20 °C). Early work by Streeter and Phelps (Streeter, 1925) determined an average value for the Ohio River of 0.23/day at 20°C (0.1/day, base 10). This value has been accepted and commonly used with reasonable results.

Specific deoxygenation rates for selected Iowa stream segments have been determined from stream surveys performed since 1977. These specific rates showed wide variations within each stream segment and among various streams. Thus, the carbonaceous deoxygenation rate of 0.2/day at 20°C is still used as an initial starting point in calibration/verification efforts. Future stream studies will be used to verify the specific rates applicable for Iowa streams.

Information on nitrogenous deoxygenation rates is extremely limited; however, available information indicates that nitrification rates (when active nitrification does occur) are somewhat greater than carbonaceous oxidation rates. Therefore, the nitrogenous deoxygenation rate (K_N) (0.3/day at 20°C was selected) is used as input data unless calibration/verification efforts provide a more reliable value. Again, future field measurements of typical nitrogenous deoxygenation rates in Iowa streams would greatly enhance the accuracy of the modeling effort.

b. Reaeration Rate Constant

Five reaeration rate constant estimation methods were provided in the spreadsheet. Each reaeration model is more accurate than others in certain circumstances. The spreadsheet gives the users the options to choose the most suitable reaeration model for a specific case.

1). The Tsivoglou & Neal (1976) model

This formulation is based on the premise that the reaeration capacity of nontidal 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.

$$K_2 = C \times S \times V$$

Where :

K_2 = Reaeration rate constant, base e, day $^{-1}$

S = Stream bed slop, 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 cms), and 15,308 for stream flow between 15 to 3,000 cfs (0.4247 to 84.95 cms)

2). Owens et al. (1964) model

This formulation is also called Owen-Gibbs and applies for stream velocity in the range of 0.1 to 5.0 fps and stream water depth in the range of 0.4 foot to 11 feet.

$$K_2 = 5.32 \times \frac{V^{0.67}}{H^{1.85}}$$

Where :

H = Stream water depth, m

3). O'Connor & Dobbins (1958) model

This formulation is more accurate when applied to moderately deep to deep channels. The suitable water channel depth should be in the range of 1 foot to 30 feet with a velocity range from 0.5 fps to 1.6 fps.

$$K_2 = 3.93 \times \frac{V^{0.5}}{H^{1.5}}$$

4). USGS (Pool-riffle) Melching and Flores (1999) model

Two formulations are included in this model, each is suitable for a certain stream flow range. When stream flow is less than 0.556 cms (or 19.64 cfs), the formulation is,

$$K_2 = 517 \times \frac{(VS)^{0.524}}{Q^{0.242}}$$

Where :

Q = Stream flow, m^3/s

When stream flow is greater than 0.556 cms (or 19.64 cfs), the formulation is,

$$K_2 = 596 \times \frac{(VS)^{0.528}}{Q^{0.136}}$$

5). USGS (Channel-control) Melching and Flores (1999) model

Similarly two formulations are included in this model, each is suitable for a certain stream flow range. When stream flow is less than 0.556 cms (or 19.64 cfs), the formulation is,

$$K_2 = 88 \times \frac{(VS)^{0.313}}{H^{0.353}}$$

When stream flow is greater than 0.556 cms (or 19.64 cfs), the formulation is,

$$K_2 = 142 \times \frac{(VS)^{0.333}}{H^{0.66} \times B_t^{0.243}}$$

Where :

B_t = the top width of the channel, m

3. <u>Temperature Corrections</u>

Temperature corrections for the carbonaceous and nitrogenous deoxygenation rate constants and the reaeration rate constants are performed within the computer model. The following equations define the specific temperature corrections used in the program:

$$K_{1(T)} = K_{1(20)} \, (1.047^{(T-20)})$$
$$K_{2(T)} = K_{2(20)} \, (1.024^{(T-20)})$$
$$K_{N(T)} = K_{N(20)} \, (1.083^{(T-20)})$$

where:

T = Water temperature, °C

The temperature corrections for the three rate constants are widely accepted formulations.

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

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

where:

T = Water temperature, °C

C_s = Saturation value for oxygen at temperature, T, at standard

Pressure of 1 atm, mg/L

4. Stream Velocity Calculations

Stream velocities are important in determining reaeration rates and the downstream dispersion of pollutants. The spreadsheet calculates velocity based on either a variation of the Manning's Formula for open channel flow or the Leopold-Maddox predictive equation.

The reality is that the field data of a lot of small streams are not available. Assumed velocity values between 0.1-0.3 fps in small streams can be used in the spreadsheet.

a. Manning's Formula

Each element in a particular reach can be idealized as a trapezoidal channel. Under conditions of steady flow, the Manning equation can be used to express the relationship between flow and depth as:

$$V = \frac{1.49 R^{2/3} S^{1/2}}{n}$$

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)(mean depth), the following equation results:

$$Q = \frac{1.49}{n} W H^{5/3} S^{1/2}$$

where:

H = Mean river depth, ft

Q = Discharge, cfs

W = Water surface width, ft

S = Slope ft/ft

n = Roughness coefficient

All variables except for "H" are input values. Internally, the program solves the above equation for H, then calculates the velocity V by:

$$V = Q/A = Q/WH$$

River slopes were obtained from existing stream profiles when available, but usually were taken from USGS topographic maps. Slopes obtained from USGS maps are rather generalized, and more accurate river profiles would greatly improve the accuracy of velocity determinations.

River widths were estimated from information obtained from field observations, flow, and cross-sectional data at each USGS gauging station.

The following table shows the roughness coefficient for various open channel surfaces. The value of 0.035 is being used on Iowa streams unless the physical characteristics of the stream are more accurately reproduced by another value.

Table 16.1. 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 at 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. Critical conditions of depth for evaluating water quality are generally much less than bankfull depth, and the relative roughness may be much higher.

In developing the particular model run for a stream segment, depth and velocity data from stream gauging 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 average length of a stream. When available, the uses of field investigations to determine actual stream velocities and depths at many selected stream sites in the modeled segment have improved the accuracy of the model.

The 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" can be calibrated. However, in most instances, more reliable flow velocity relationships can be modeled by using the power equations.

b. Power Equations

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

$$V = aQ^b$$

$$H = \alpha Q^\beta$$

where a, b, α and β are empirical coefficients that are determined from velocity-discharge and stage-discharge rating curves, respectively. The values of velocity and depth can then be employed to determine the cross-sectional area and width by

$$A_c = \frac{Q}{V}$$

$$W = \frac{A_c}{H}$$

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 that the empirical constants a and b apply to a specific stream cross section. The value of "b" represents the slope of a logarithmic plot of velocity versus discharge. The value of "a" represents the velocity at a unit discharge. The exponents b and β typically take on values listed in Table 16.2. Note that the sum of b and β must be less than or equal to 1. If this is not the case, the width will decrease with increasing flow. If their sum equals 1, the channel is rectangular.

Table 16.2. Typical values for the exponents of rating curves used to determine velocity and depth from flow (Barnwell et al. 1989).

Equation	Exponent	Typical value	Range
$V = aQ^b$	b	0.43	0.4–0.6
$H = \alpha Q^\beta$	β	0.45	0.3–0.5

The power equations have been used in many studies and have been found to produce reliable results when the empirical constants are properly evaluated. However, its use is limited to streams for which historical data are not available for determining representative values for the empirical constants. A regression analysis can be performed on several sets of velocity-discharge data to determine the empirical constants. The data selected for use in the analysis corresponds to low stream flow conditions since the use of elevated stream flow data may bias the results.

Since reaches of uniform cross section, slope, and roughness parameters rarely characterize stream systems, the empirical constants are determined for several representative cross sections of each stream system to be modeled. 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 can also be obtained from the USGS gauging station data or from stream surveys.

QUALIIK Model

QUAL2K is a river and stream water quality model that is intended to represent a modernized version of the QUAL2E model (Brown and Barnwell 1987).

QUAL2K is similar to QUAL2E in the following respects:

- One dimensional. The channel is well-mixed vertically and laterally.
- Steady state hydraulics. Non-uniform, steady flow is simulated.
- Diurnal heat budget. The heat budget and temperature are simulated as a fuction of meteorology on a diurnal time scale.

- Diurnal water-quality kinetics. All water quality variables are simulated on a diurnal time scale.
- Heat and mass inputs. Point and non-point loads and abstractions are simulated.

The QUAL2K framework includes the following new elements:

- Software Environment and Interface. Q2K is implemented within the Microsoft Windows environment. Numerical computations are programmed in Fortran 90. Excel is used as the graphical user interface. All interface operations are programmed in the Microsoft Office macro language: Visual Basic for Applications (VBA).
- Model segmentation. QUAL2E segments the system into river reaches comprised of equally spaced elements. QUAL2K also divides the system into reaches and elements. However, in contrast to QUAL2E, the element size for Q2K can vary from reach to reach. In addition, multiple loadings and withdrawals can be input to any element.
- Carbonaceous BOD speciation. QUAL2K uses two forms of carbonaceous BOD to represent organic carbon. These forms are a slowly oxidizing form (slow CBOD) and a rapidly oxidizing form (fast CBOD).
- Anoxia. QUAL2K accommodates anoxia by reducing oxidation reactions to zero at low oxygen levels. In addition, denitrification is modeled as a first-order reaction that becomes pronounced at low oxygen concentrations.
- Sediment-water interactions. Sediment-water fluxes of dissolved oxygen and nutrients can be simulated internally rather than being prescribed. That is, oxygen (SOD) 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 waters.
- Bottom algae. The model explicitly simulates attached bottom algae. These algae have variable stoichiometry.
- Light extinction. Light extinction is calculated as a function of algae, detritus and inorganic solids.
- pH. Both alkalinity and total inorganic carbon are simulated. The river's pH is then computed based on these two quantities.
- Pathogens. A generic pathogen is simulated. Pathogen removal is determined as a function of temperature, light, and settling.
- Reach specific kinetic parameters. QUAL2K allows you to specify many of the kinetic parameters on a reach-specific basis.
- Weirs and waterfalls. The hydraulics of weirs as well as the effect of weirs and waterfalls on gas transfer are explicitly included.

A detailed documentation and User's Manual for QUALIIK water quality model can be obtained from the EPA website. The User's Manual provides a 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. While running the program for a specific stream or for calibrating a segment, the suggested ranges for reaction coefficients are presented in Table 16.3. These values serve as a guide for a run of the QUAL-IIK program.

TABLE 16.3

RECOMMENDED RANGES FOR REACTION COEFFICIENTS

FOR QUAL-IIK

DESCRIPTION	UNITS	RANGE OF VALUES
Ratio of chlorophyll-a to algae biomass	ug Chl-a/Mg A	10 - 100
Fraction of algae biomass that is nitrogen	Mg N/Mg A	0.07 – 0.09
Fraction of algae biomass that is phosphorus	Mg P/Mg A	0.01 – 0.02
O_2 Production per unit of algal growth	Mg O/Mg A	1.4 – 1.8
O_2 Uptake per unit of algae respired	Mg O/Mg A	1.6 – 2.3
O_2 Uptake per unit of NH_3 oxidation	Mg O/Mg N	3.0 – 4.0
O_2 Uptake per unit of NO_2 oxidation	Mg O/Mg N	1.0 – 1.14
Rate constant for the biological oxidation of NH_3 to NO_2	1/Day	0.10 – 1.00
Rate constant for the biological oxidation of NO_2 to NO_3	1/Day	0.20 – 2.00
Rate constant for the hydrolysis of organic-N to ammonia	1/Day	0.02 – 0.4
Dissolved phosphorus removal rate	1/Day	0.02 – 0.4
Organic phosphorus settling rate	1/Day	0.001 – 0.10
Algal settling rate	ft/Day	0.5 – 6.0
Benthos source rate for phosphorus	Mg P/day-ft	Highly Variable
Benthos source rate for NH_3	Mg N/day-ft	Highly Variable
Organic P decay rate	1/Day	0.1 – 0.7
Carbonaceous deoxygeneration rate constant	1/Day	0.02 – 3.4

Reaeration rate constant	1/Day	0.0 - 100

RECOMMENDED RANGES FOR REACTION COEFFICIENTS

FOR QUAL-II

- Continued -

DESCRIPTION	**UNITS**	**RANGE OF VALUES**
Rate of loss of CBOD due to settling	1/Day	-0.36 to 0.36
Benthic oxygen uptake	Mg O/day-ft	Highly Variable
Coliform die-off rate	1/Day	0.5 – 4.0
Maximum algal growth rate	1/Day	1.0 – 3.0
Algal death rate	1/Day	0.024 – 0.24
Preferential NH_3 uptake factor	--------	0.0 – 0.9
Algal N to organic N decay rate	1/Day	0.11
Algal respiration rate	1/Day	0.05 – 0.5
Michaelis-Menton half-saturation constant for light	Langleys/min	0.02 – 0.10
Michaelis-Menton half-saturation constant for nitrogen	mg/l	0.01 – 0.20
Michaelis-Menton half-saturation constant for phosphorus	mg/l	0.01 – 0.05
Non-algal light extinction coefficient	1/ft	Variable
Algal light extinction coefficient	(1/ft)/(ug Chl-a/L)	0.005 – 0.02

MODELING DATA SOURCES

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 may be used in stream modeling for wasteload allocations.

Wastewater Discharges

The required data for each discharger consists of effluent flow rates and effluent characteristics such as Biochemical Oxygen Demand (BOD), ammonia nitrogen (NH_3-N), Dissolved Oxygen (DO) concentrations, and temperature. The specific location and characteristics of some smaller wastewater discharges are often unknown and are determined from field investigations or during special stream surveys. Most wastewater discharge information is available in the departmental files.

River Miles

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 must be established and river miles need to be located such that the location of tributaries, etc., can be determined. The best maps to start with are U.S.G.S. topographic maps. These consist of section maps (scale: 1:250,000) and quadrangle maps (scale: 1:24,000). Other maps such as state and county road maps can also be used to supplement the U.S.G.S. maps.

Field Reconnaissance

The following data can be 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 should be performed, if possible, during flow conditions that represent the flows used in the modeling effort. Stream discharge information during stream surveys may be verified from data obtained from the U.S.G.S. The stream flow observed during stream surveys is often greater than the $7Q_{10}$. Data such as river widths need to be extrapolated downward to represent $7Q_{10}$ conditions. Shapes of channel cross sections are an aid in this determination.

River Channel Slopes

After river miles and locations are established, the next step is the determination of river channel slopes. During low flow conditions it can be assumed that river channel slopes are essentially the same as the slope of the water surface. Channel profiles can be used as representative of water surface slopes. In some cases, profiles of the river have already been determined. The U.S. Army Corps of Engineers usually does this as part of the work conducted prior to proposal or construction of flood control reservoirs. Without accurate profiles, river slopes can be determined from U.S.G.S. contour maps 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. U.S.G.S. quadrangle maps (if available) are a more reliable source of slope data. A GIS elevation coverage can also be used to obtain the stream slopes. Often, these are the only sources available and are the best method of slope determination without an extensive field survey.

River Widths and Roughness Coefficients

River widths and roughness coefficients can be estimated during the field reconnaissance. Roughness coefficients can also be estimated from Table 16.1 above.

The variation of river widths with discharges can often be determined from data at U.S.G.S. gauging stations. The U.S.G.S. periodically calibrates each gauge. The results from these calibrations are available on U.S.G.S. website and include widths, cross-sectional area, mean velocities, and discharges. Reasonably accurate estimations of river widths at the desired discharge can usually be made with this gauging station information along the river widths measured during special stream surveys.

Stream Flow

In the determination of flow conditions throughout the river system to be modeled, all available data from U.S.G.S. flow measuring stations as well as flow rates from all of the wastewater discharges must be obtained. River flows need to be allocated among tributary, groundwater, and wastewater inflow sources. The design low flow is used as the modeling basis, and is determined from a statistical analysis of the flow records at each of the gauging stations in the river system. Design low flows have already been determined for partial and continuous gauging stations (i.e. Statistical Summaries of Selected Iowa Stream flow Data Through September 1996" by USGS and Iowa Natural Resources Council, Annual and Seasonal Low-Flow Characteristics of Iowa Streams, Bulletin No. 13, 1979). The design low flows at gauging stations must then be allocated to tributaries based on drainage areas. Tributary drainage areas may be available from existing publications (i.e. Larimer, O.J., Drainage Areas of Iowa Streams, Iowa Highway Research Bulletin No. 7, 1957) or they can be determined from U.S.G.S. contour maps. The Department staff uses a GIS tool to estimate the drainage areas and the corresponding stream critical low flows.

A summation of tributary inflows and wastewater discharges often is less than the gauged flow. The difference is usually distributed along the main channel of the river as a uniform inflow in terms of cfs per mile of river reach length. If the gauged 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.

<u>Tributary and Groundwater Quality</u>

Values for BOD, NH_3-N, and DO of tributary and groundwater inflow are required for stream modeling. Often, a main tributary to the stream being modeled has also been modeled. In this case, the water quality of the tributary just before discharge into the main stream (as determined by the model) is used. If the tributary is small and has several wastewater discharges, hand calculations can be done to determine its water quality just before entering the main stream.

If the tributary is free of continuous discharging wastewater facilities, water quality has been assumed to be good. The tributary water quality input values are: ultimate BOD – 6 mg/l; NH_3-N concentrations – 0.0 mg/l (summer), 0.5 mg/l (fall, winter, and spring); and DO at saturation.

Groundwater is also noted to be of high quality. The model input values for groundwater are ultimate BOD of 6 mg/l and NH_3-N at 0 mg/l. Groundwater DO's may be quite low depending on how it enters the stream. If it is subsurface flow, DO may be close to zero. A groundwater DO of 2 mg/l is used in wasteload allocation (WLA) work in Iowa.

<u>Rate Constants</u>

The reaeration rate constant (K_2) is usually determined from one of many available predictive formulas shown in the previous section. The document titled 'Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling – EPA/600/3-85/040, June 1985" can be a good source for obtaining the initial values for rate constants.

Carbonaceous and nitrogenous deoxygenation 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 may be used. In most cases the carbonaceous deoxygenation rate constant (K_1) will not be less than 0.2 per day (20°C). Values as high as 3.4 per day (20°C) have been reported in the literature.

Less information is available on the nitrogenous deoxygenation rate constants or nitrification rates in streams. Experimental work in Illinois (State of Illinois, Environmental Protection Agency, <u>Guidelines for Granting of Exemptions from Rule 404(C) and 404(F) Effluent Standards</u>, Oct., 1974) determined that the nitrogenous deoxygenation rate constant (K_N) ranged from 0.25 to 0.37 per day with an average value of 0.29 per day at 20°C. The current model uses a K_N value based on stream calibration from the modeled stream or similar streams. Other rate constants for benthic and algal kinetics are based on calibration data or literature values. Specific explanations of these rate constants are in the User's Manual for the QUALIIK model.

Dams and Impoundments

The damming of a stream creates special conditions for water quality modeling. For modeling purposes, dams and the resulting impoundments can be put into one of two classifications.

1. Large dams that back up rather extensive impoundments. Flow through the impoundment is not "plug flow" and inflow may be dispersed in a variety of vertical and horizontal directions.
2. Low-head dams which essentially make the river channel wider and deeper for a maximum distance of several miles. Flow through the impoundment is primarily "plug flow."

Class 1 dams and impoundments cannot easily be modeled to predict water quality. The modeling effort should be stopped at the beginning of the impoundment and started again below the dam. Water quality below the dam can be estimated from knowledge of the size of the impoundment, the method of water withdrawal, and water quality data from stream surveys. Water taken from the lower levels of an impoundment during periods of summer stratification may be low in DO. If water flows over a spillway or an overflow weir it may be close to the DO saturation point. One can expect the BOD and NH_3-N concentrations in the discharge from large impoundments to be low unless the impoundment is highly eutrophic.

Class 2 dams and impoundments can be modeled by treating the impoundment as an enlarged or slower moving reach of the river. The length of the pool backed up by the dam may be divided into one or more reaches. Widths can be approximated from field observations. Slopes are taken as the water surface elevation and are quite small, generally elevation drops off no more than a foot over the length of the pool.

The dams may be treated as a reach 0.001 miles or 5.28 feet in length. The slope of this reach then becomes the dam height divided by 5.28 feet. The only water quality parameter that is significantly affected through the dam reach is the DO. Tsivoglou's reaeration rate constant prediction formula can be used to quite effectively predict reaeration over a dam. The equation for change in the DO deficit with time is:

$$D_t = D_o e^{-K_2 t}$$

where:

D_t = DO deficit at time, t

D_o = DO deficit at time zero

K_2 = Reaeration rate constant

Tsivoglou's reaeration rate constant predictive equation (neglecting ice conditions) is:

$$K_2 = \frac{c\Delta H}{t}$$

where:

c = Escape coefficient

ΔH = Change in elevation in time, t

Substituting into the DO deficit equation one obtains:

$$D_t = D_o e^{-c\Delta H}$$

Example:

With a dam 10 feet high and c = 0.115/ft. the ratio of D_t/D_o is 0.32 or the deficit is 32 percent of the deficit at time zero. This is a DO deficit recovery of 68 percent.

QUAL2K includes a component that can estimate the effect of control structures on oxygen and it is described as follows:

Oxygen transfer in streams is influenced by the presence of control structures such as weirs, dams, locks, and waterfalls. Butts and Evans (1983) have reviewed efforts to characterize this transfer and have suggested the following formula,

$$r_d = 1 + 0.38 a_d b_d H_d (1 - 0.11 H_d)(1 + 0.046T) \qquad \textbf{(16.1)}$$

where r_d = the ratio of the deficit above and below the dam, H_d = the difference in water elevation [m], T = water temperature (°C) and a_d and b_d are coefficients that correct for water-quality and dam-type. Values of a_d and b_d are summarized in Table 7 – coefficient values used to predict the effect of dams on stream reaeration. QUAL2K manual. If no values are specified, QUAL2K uses the following default values for these coefficients: a_d = 1.25 and b_d = 0.9.

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