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State of Washington

Appendices

Water Quality Program
Permit Writer's Manual

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Appendices

Water Quality Program
Permit Writer's Manual

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Appendix B

1. Autocorrelation

The background material for this example is presented in Chapter 4, Part 4.

1.1 Adjustment of Monthly Average Effluent Limitations for Autocorrelation

The Washington State Department of Ecology (Ecology) required an industrial discharger to monitor its cyanide discharge for several years prior to permit renewal. The data are presented below in Table B-1. The data indicated a reasonable potential for violation of the water quality standards and the necessity of a water quality-based effluent limit in the new permit. The facility requested an adjustment of the average monthly effluent limit due to autocorrelation. The request and data were passed on to the U.S. Environmental Protection Agency (EPA) for assistance in evaluating the autocorrelation. Calculation of autocorrelation involves two steps: (1) calculation of autocovariance and then (2) using the autocovariance to calculate the significant autocorrelation coefficients. These coefficients are used in calculating the monthly average effluent limitation.

An example is presented to show the process but the example shows a one-week cycle and would probably not be granted an adjustment to the effluent limit. The facility used in this example applied for an autocorrelation adjustment but also provided the data for a higher site-specific water quality criteria for cyanide. With the higher criteria, they no longer showed a reasonable potential or required an effluent limit.

Table B-1. Effluent cyanide data

Date	Cyanide ug/l	Date	Cyanide ug/l	Date	Cyanide ug/l
3/2/90	38	7/3/90	15	12/16/93	79
3/3/90	70	7/5/90	10	12/22/93	38
3/4/90	50	7/10/90	11	12/29/93	47
3/5/90	66	7/12/90	10	01/05/94	27
3/6/90	52	7/17/90	3	01/12/94	33
3/7/90	35	7/19/90	5	01/19/94	28
3/14/90	40	7/24/90	3	01/26/94	42
3/15/90	47	7/26/90	4	02/02/94	29
3/22/90	3	8/1/90	5	02/09/94	37
3/23/90	23	8/2/90	5	02/16/94	87
3/27/90	2	8/7/90	5	02/23/94	44
3/30/90	18	8/16/90	3	03/01/94*	70
4/3/90	35	8/20/90	4	03/02/94	45
4/4/90	30	8/23/90	4	03/09/94	43
4/5/90	32	8/28/90	3	04/06/94	20
4/6/90	34	8/30/90	5	04/13/94	12
4/9/90	39	4/1/91	60	04/19/94	15
4/12/90	44	4/8/91	10	04/27/94	26
4/17/90	48	03/26/93	60	05/04/94	22
4/19/90	51	03/30/93	56	05/11/94	23
4/25/90	64	04/02/93	58	05/18/94	16
4/26/90	33	04/06/93	40	05/25/94	21
4/29/90	70	04/08/93	74	06/01/94	19
4/30/90	34	04/12/93	66	06/08/94	17
5/8/90	15	04/19/93	52	06/15/94	15
5/10/90	13	04/22/93	37	06/30/94	23
5/15/90	20	04/25/93	39	07/06/94	20
5/17/90	20	05/10/93	21	07/13/94	15
5/22/90	23	05/12/93	13	7/20/94	11
5/24/90	18	05/14/93	12	7/27/94	10
5/29/90	8	6/1/93*	20	8/3/94	14
5/31/90	12	06/16/93	13	8/10/94	10
6/5/90	10	06/18/93	11	8/17/94	4
6/7/90	2	06/21/93	9	8/24/94	13
6/12/90	2	06/24/93	13	8/31/94	8
6/14/90	2	07/10/93	19	9/7/94	35
6/19/90	23	07/12/93	18	9/14/94	26
6/21/90	17	07/14/93	12	9/21/94	38
6/26/90	13	07/16/93	15	9/28/94	46
6/28/90	14	07/29/93	20	10/5/94	5
		08/04/93	13	10/11/94	6

Date	Cyanide ug/l
10/18/94	4
10/26/94	4
11/2/94	35
11/9/97	44
11/17/94	34
11/22/94	35
11/30/94	26
12/7/94	43
12/14/94	53
12/21/94	30
12/28/94	34
1/5/95	34
1/11/95	53
1/26/95	46
2/1/95	41
2/8/95	49
2/15/95	24
2/22/95	19
3/1/95	32
3/8/95	33
3/15/95	22
3/29/95	23

Calculation of Autocovariance Estimates

The cyanide data consists of 144 individual daily composite samples representing concentration measurements taken during the period March 2, 1990 to March 29, 1995. Days with no samples are considered missing observations. For purposes of calculating the autocovariances, let z_1, z_2, ..., z_N denote the N concentration measurements. The following formula (Box & Jenkins, 1976, p.32) was used to calculate the estimate of the autocovariance:

$$C_k = \frac{1}{N}\sum_{t=1}^{N-k}(Z_t - \overline{Z})(Z_{t+k} - \overline{Z})$$

where,

N = number of observations = 144
k = lag between observations in days = 0, 1,...,30
t = the day, where
t = 1 = March 2, 1990
t = 2 = March 3, 1990
t = 3 = March 4, 1990
t = 1854 = March 29, 1995

Days on which concentration amounts were not reported are considered to be missing observations. Therefore only those values of t where data has been reported will be used to calculate autocovariances.

The following lists t=1 through t=13 and gives some calculation examples:

value of t	date	concentration amount
1	3/2/90	38
2	3/3/90	70
3	3/4/90	50
4	3/5/90	66
5	3/6/90	52
6	3/7/90	35
7 – 12	3/8/90 - 3/13/90	missing
13	3/14/90	40

The mean of the concentration amounts for the entire data set is:

$$\overline{Z} = \frac{1}{N} \sum_{t=1}^{N-k} Z_t = 26.9653$$

Note: Only use those values of t where concentration amounts exist.

Calculations for a Lag of One Day Using All Measurements One Day Apart (i.e., k=1).

For example:

value of t	calculation
1	(38-26.9653)(70-26.9653) = 474.875
2	(70-26.9653)(50-26.9653) = 991.291
3	(50-26.9653)(66-26.9653) = 899.153
4	(66-26.9653)(52-26.9653) = 977.222
6 - 13	*
•	•
•	•
•	•
1853	*
N = 144	Sum of contributions = 5759.7696

Autocovariance of lag two = 8784.0864/144 = 61.0006

* concentration amount two days from this value of t is missing and therefore cannot contribute toward an autocovariance for a lag of two. Concentration values two days apart must exist to be included in the calculation of the autocovariance for lag two.

It is recommended (Box & Jenkins, 1976, p. 33) that k not be larger than N/4. In our case, we recommend that k not be larger than 30 since the limits are calculated on a monthly basis. The estimate of the *k*th lag autocorrelation is

$$r_k = \frac{C_k}{C_o}$$

The number of observations used to estimate r_k and c_k need not be equal. Therefore missing data are not a problem; however, missing data does prevent standard time series modeling. Estimates of autocovariance and autocorrelation for lags 1-30 are given in Table B-2.

A lag of 7 indicates the autocorrelation for observations 7 days apart. (For example, successive Mondays would have a lag of 7, successive Tuesdays would have a lag of 7.)

Table B-2. Estimates of Autocovariance and Autocorrelation: Lags 0-30

Lag	Autocovariance	Autocorrelation
0	367.117	
1	39.9984	0.10895
2	61.0006	0.16616
3	33.7574	0.09195
4	44.5250	0.12128
5	37.6707	0.10261
6	49.7120	0.13541
7	101.503	0.27649
8	27.0599	0.07371
9	39.8864	0.10865
10	28.6923	0.07816
11	16.0746	0.04379
12	32.0776	0.08738
13	61.6921	0.16804
14	75.6883	0.20617
15	8.7124	0.02373
16	18.3915	0.05010
17	16.0578	0.04374
18	-4.7897	-0.01305
19	17.2813	0.04707
20	9.6339	0.02624
21	68.9943	0.18794
22	0.4231	0.00115
23	19.9909	0.05445
24	-0.0996	-0.00027
25	-1.4486	-0.00395
26	18.0623	0.04920
27	9.7315	0.02651
28	57.6594	0.15706
29	-0.4159	-0.00113
30	14.6503	0.03991

Significance of Autocovariances

Using

$$\text{var}[r_k] = \frac{1}{N}$$

to approximate the variance of the autocorrelations (Box & Jenkins, 1976, p. 35), where N = 144, the approximate variance is 0.006944. The corresponding approximate standard error is 0.083307. The approximate standard error is compared to the autocorrelations (Box & Jenkins, 1976, p .36). In cases where the autocorrelation is greater than two standard deviations (0.166614) the autocovariance was considered significant. Autocovariances for lags 2, 7, 13, 14, and 21 were significant. Adjustment for the lag 2 autocorrelation was not made in our limitation calculations because daily monitoring would be required to implement such limits properly and the permit writer had indicated a requirement of weekly monitoring.

Calculation of Limits

Calculation for limits using the covariance of lags 7, 14, and 21 days follows:

$$\begin{aligned}\text{Var}[\text{Mean}(X)] &= (1/16)[4 * \text{Var}(X) + 6 * 0.27649\ \text{Var}(X) + \\ &\quad 4 * 0.20617\ \text{Var}(X) + 2 * 0.18794\ \text{Var}(X)]\\ &= .25\ \text{Var}(X) + 0.10368\ \text{VAR}(X) + 0.05154\ \text{Var}(X)\\ &\quad + 0.02349\ \text{Var}(X)\\ &= 0.42871\end{aligned}$$

1. Performance-based Limit
   a. Calculate μ and σ as above as the mean and standard deviation of the ln(CN) values.

$$\mu_y = 2.96442$$
$$\sigma_y = 0.91308$$
$$\sigma_y^2 = 0.83372$$

b. Calculate the estimated mean and standard deviation of the CN values as:

$$\begin{aligned}E(X) &= \exp(\mu_y + \sigma_y^2/2)\\ &= \exp(2.96442 + 0.83372/2)\\ &= \exp(3.38128)\\ &= 29.40839\end{aligned}$$

$$\begin{aligned}\text{Var}(X) &= \exp(2 * \mu_y + \sigma_y^2) * [\exp(\sigma_y^2) - 1]\\ &= \exp(2 * 2.96442 + 0.83372)\\ &\quad * [\exp(0.83372) - 1]\end{aligned}$$

$= \exp(6.76256) * [\exp(0.83372) - 1]$
$= (864.85339) * (1.30187)$
$= 1125.9266$

c. Calculate the estimated mean and variance of the average of 4 weekly CN values used in a monthly average. Include the autocovariance effect.

$n = 4$
$\sigma_4^2 = \ln \{0.42871 * Var(X) / E[(X)]^2 + 1\}$
$= \ln \{(0.42871 * 1126 / [29.4]^2) + 1\}$
$= \ln 1.55848$
$= 0.44371$

$\mu_4 = \ln(E(X)) - 0.5\sigma_4^2$
$= \ln(29.4) - 0.5 * (0.44371)$
$= 3.38099 - 0.22186$
$= 3.15913$

d. Calculate the 95th percentile monthly average limit

$X_{.95} = \exp\{\mu_4 + 1.645\sigma_4\}$
$= \exp\{3.15913 + 1.645\ (0.66611)\}$
$= \exp\{4.25488\}$
$= 70.44836$

e. Calculate the 99th percentile daily average limit

$X_{.99} = \exp\{\mu_y + 2.326\sigma_y\}$
$= \exp\{2.96442 + 2.326(0.91308)\}$
$= \exp\{5.08824\}$
$= 162.10496$

2. Calculating the limit based on the proposed site specific acute water quality standard for CN of 9.85μg/l.

a. WLA = (9.85 μg/l)(13) = 128 μg/1

b. Calculate the LTA using the CV calculated from the lognormal distribution

$CV = [\exp(\sigma_y^2) - 1]^{0.5}$
$= [\exp(0.83372) - 1]^{0.5}$

$= 1.30187]^{0.5}$
$= 1.14100$

$\sigma_2 = \ln(CV^2 + 1)$
$= \ln(1.14100^2 + 1)$
$= \ln(2.30188)$
$= 0.83373$

$LTA = WLA \{\exp[0.5\sigma^2 - z\sigma]\}$
$= 128 \{\exp[0.5(0.83373) - 2.326(0.91309)]\}$
$= 128 \{\exp[-1.70698]\}$
$= 128 \{0.18141\}$
$= 23.22048$

c. Calculate the MDL and the AML

$MDL = LTA \{\exp[z\sigma - 0.5\sigma^2]\}$
$= 23.22048 \{\exp [2.326(0.91309) - 0.5(.83373)]$
$= 23.22048 \{\exp [1.70698]\}$
$= 23.22048 \{5.51229\}$
$= 127.99801$

$AML = LTA \{\exp[z\sigma_4 - 0.5\sigma_4^2]\}$
$\sigma_4^2 = \ln [(0.42871)CV^2 + 1]$
$= \ln [(0.42871)(1.14100)^2 + 1]$
$= \ln [1.55813]$
$= 0.44349$

$AML = 23.22048 \{\exp[(1.645)(0.66595) -$
$(0.5)(0.44349)]$
$= 23.22048 \{\exp[0.87375]\}$
$= 23.22048 \{2.39588\}$
$= 55.63348$

Table B-3 illustrates the different results obtained when considering no autocorrelation, autocorrelation at lag 7, autocorrelation at lags 7 and 14, and autocorrelation at lags 7, 14, and 21. The limits increase with the amount of autocorrelation included in the calculation. Positive autocorrelation increases the variance which increases the limit.

Table B-3. Comparison of Limit

Lags Considered Significant	Performance-Based AML	Water Quality-Based AML	σ_4^2
None (Independent)	47.3	48.5	.282
Lag 7	66.9	52.8	.378
Lags 7 and 14	69.4	54.8	.424
Lags 7,14,21	70.5	55.6	.444

Note: MDLs are the same no matter how many lags are considered significant. For the Performance based limit it would be 159.3 and the water quality-based limit is 128

2. An Alternative Method for Estimating Upper Quantiles

This method estimates a selected upper percentile value from a distribution assumed to be lognormally distributed. The most statistically valid estimate of an upper percentile value is a maximum likelihood estimator which is proportional to the population geometric mean. If one assumes the population of effluent concentrations to fit a lognormal distribution, this relationship is given by:

$C_p = C_{mean} * \exp(Z_p * \sigma - 0.5 * \sigma^2)$

where: Z_p = normal distribution factor at pth percentile
$\sigma^2 = \ln(CV^2 + 1)$

To calculate the maximum likelihood estimator of the 95th percentile, the specific relationship becomes:

$C_{95} = C_{mean} * \exp(1.645 * \sigma - 0.5 * \sigma^2)$

if CV is assumed = 0.6,
$\sigma^2 = .307$

The ratio of the estimated 95th percentile value to the mean (C_{95}/C_{mean}) = 2.13

A single effluent value or the geometric mean of a group of values is multiplied by the ratio to yield the estimate of the 95th percentile value.

The following table shows the ratio of the upper percentile to the mean for the 90th, 95th, and 99th percentiles

2.1 Ratio of Upper Percentiles to Geometric Mean

Percentile	Z	C_p/C_{mean}
90	1.283	1.74
95	1.645	2.13
99	2.386	3.11

In use with limited data sets assumed to be lognormally distributed, the geometric mean is multiplied by the value in the right column above to estimate the percentile given in left column. This estimation technique results in lower estimates of upper percentile values than the technique discussed in the TSD in Section 3.2.2. for *n* less than 6. At some number of values, it becomes more accurate simply to calculate the desired percentile value. Most spreadsheets have this capability. The number of values at which this occurs can't be predicted because it depends on the characteristics of the population being sampled. Using the TSD upper quantile estimation technique on page 56 and calculating where the largest value is greater than the 90th percentile (at 95% Confidence), the value of *n* is 30. In other estimation techniques for lognormal distributions, Gilbert (1987) predicts a large sample as *n* = 20.

3. Deriving the 7Q10 High Flow (HF) and the Design Spill

3.1 Guidelines for Deriving the 7QHF

Use the record of observed flow data. For the Columbia River use the record from water year 1974 to the present. 1974 is the year when the last major storage reservoir was built in the Columbia River Basin. If data are not available for the location of interest, make reasonable calculations based on an approved 7Q10hf for up-river or down-river locations or dams. Flow data must include total river flow.

The period of record may be extended by including modeled or transformed data prior to 1974 that represents the current condition of the basin with all dams in place. If this is done, compare results of using data from the observed period of record with the results of using the data from the extended period of record. This serves as a check to see if the proposed method for extending the period of record gives comparable results.

To the extent possible, take into account any trends or anticipated changes of flows in the future.

Use daily average flows to calculate the 7Q10hf and then determine the highest 7-consecutive-day average peak flow for each year.

Calculate frequency/return interval using standard hydrology methods (see USGS bulletin 17B, *Guidelines for Determining Flood Flow Frequency*, or any hydrology textbook)

Information submitted to Ecology must include documentation of the methods of calculating 7Q10hf, including assumptions about data, current trends, anticipated changes, quality assurance, methods of measurement, methods of transforming historic data and comparisons of 7Q10hf values for other dams.

3.2 Determining Design Spill for Gas Abatement

To determine the design spill for gas abatement, use the hourly spill data for high flow months of a high flow year (1997 for example) and extrapolate to the 7Q10hf. The high flow months for the Columbia River are May and June.

Appendix C

Guidance for Conducting Mixing Zone Analyses

by
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1. Introduction

Washington Administrative Code (WAC) 173-201A-400 contains regulations on mixing zones in the State of Washington. This document contains guidelines on how these regulations should be implemented.

The key products from a mixing zone analysis are the dilution factors. They are used in conjunction with the water quality criteria (WAC 173-201(A)) for calculating reasonable potentials and effluent limits, and for evaluating whole effluent toxicity (WET) characterization of effluent and deriving WET effluent limits (WAC 173-205). These effluent limits are then incorporated into a NPDES (National Pollutant Discharge Elimination System) permit issued by the Department authorizing a point source to discharge into waters of the state.

There are aquatic life-based water quality criteria and human health-based water quality criteria. The former are applied at both the acute and chronic boundaries; the latter are applied at the chronic boundary. The processes for conducting aquatic life-based analyses and human health-based analyses parallel each other (Chapter 6 and 7 of Ecology's *Permit Writer's Manual*: https://fortress.wa.gov/ecy/publications/summarypages/92109.html). The differences are in the choice of mixing zone boundaries, and the selection of reasonable worst-case versus average values for the various parameters used in the analyses - as explained later in this document. The permit manager should be consulted about the need for a human health-based analysis.

Steady-state models are the most frequently used tools for conducting mixing zone analyses. However, dynamic modeling is gaining some popularity as the inherent intensive computations become faster with powerful computers. In some circumstances the primary tool may be a dye study - with a model filling a secondary role. One such circumstance would be when it is apparent that an effluent plume does not develop normally (for any number of reasons). The dilution factors must then be measured directly in the field. But, they are the dilution factors for one set of effluent and receiving water conditions only, and a model may still be necessary for analyzing other sets of conditions that are quite different from those present during the dye study. The most appropriate model to use will be the one that validates best against the dye results.

This guidance provides the specific, detailed information that is needed to select the correct values for the effluent and receiving water parameters, select the appropriate model, and determine when a dye study should be used. It is not a standalone user's manual or a “cookbook”. It is essential to have a working knowledge of how water quality-based effluent limits are developed in Washington State. This knowledge can be gained through reading and understanding the *Water Quality Standards for Surface Waters of the State of Washington* (in particular the subparts on **Toxic Substances** and **Mixing Zones**) and the Department of Ecology’s *Permit Writer’s Manual* (in particular Chapters 6 and 7) - and through experience.

This guidance fills in most of the knowledge gaps so that consultants and permit managers will be able to operate and communicate from the same, uniformly high-level of understanding and expertise needed to produce quality products. Placing this guidance in the *Permit Writer’s Manual* and on the Internet (http://www.ecy.wa.gov/programs/eap/mixzone/mixzone.html) ensures that it is a living document that is continually updated as more experience and feedback occurs. As with the *Permit Writer’s Manual,* it is expected that ample justification will be provided whenever the guidance is not followed.

2. Mixing Zone General Restrictions

WAC 173-201A-400(2) requires that a discharger is required to fully apply AKART (all known, available, and reasonable methods of treatment) prior to being authorized a mixing zone. A statement that AKART is being fully applied by the discharger must accompany any document used to seeking a mixing zone from Ecology.

WAC 173-201A-400(4) requires that no mixing zone shall be granted unless the supporting information clearly indicates the mixing zone would not:

a. have a reasonable potential to cause a loss of sensitive or important habitat,
b. substantially interfere with the existing or characteristic uses of the water body,
c. result in damage to the ecosystem, or
d. adversely affect public health as determined by Ecology.

A section should be included in any document seeking a new mixing zone from Ecology that addresses the environmental threats discussed above. An “environmental” map should be included that shows the spatial location of the proposed outfall and mixing zones in relation to the water body reach and locations of any sensitive habitat, water supply intakes, recreational areas, shellfish areas, etc.

WAC 173-201A-400(5) requires that water quality criteria shall not be violated outside of the boundary of a mixing zone as a result of the discharge for which the mixing zone was authorized. This also means that if the ambient waters already exceed the water quality criteria, no mixing zone can be allowed. However, an exception to this rule would be when a waterbody’s temperature is warmer than the criteria and the condition is due to natural conditions, then a 0.3° C allowance over background is applicable as per WAC 173-201A-210(1)(c)(i). In this case a mixing zone is allowed which would allow the ambient temperatures to exceed by 0.3° C at the edge of the mixing zone.

WAC 173-201A-400(7)(d) includes restrictions for mixing zones in lakes. Mixing zones in lakes are discouraged unless it is demonstrated that discharge to the lake is unavoidable, that there is an overriding public interest and AKART is fully implemented as outlined in (i) to (iii). Any requests for a mixing zone in a lake must accompany an analysis of these conditions.

WAC 173-201A-400(9)(a) (ii) requires that overlapping mixing zones may only be allowed if the combined effects would not create a barrier to the migration or translocation of indigenous organisms to a degree that has the potential to cause damage to the ecosystem. For overlapping mixing zones, such an analysis should be presented when requesting mixing zone from Ecology.

WAC 173-201A-400(12), WAC 173-201A-400(13) and WAC 173-201A-400(14) contain requirements for any proposed exceedance of the numeric size of mixing zones described in WAC173-201A-400(7). This includes regular mixing zones, overlapping mixing zones (WAC173-201A-400(9)), stormwater (WAC173-201A-400(10)), and combined sewer overflows (WAC173-201A-400(11)).

3. What is a Dilution Factor

If Q_a is the volume flux of ambient water entrained in the discharge plume from an outfall at some distance; and Q_e is the volume flux of effluent in the plume, then the dilution factor DF is defined as:

$$DF = \frac{(Q_a + Q_e)}{Q_e} \tag{1}$$

Q_a is extremely difficult to quantify. However, what can be measured directly in the plume is the concentration of a pollutant (or a dye).

If C_p is the plume concentration of a pollutant (or a dye) and C_a and C_e are the ambient and effluent concentrations of the pollutant (or a dye), respectively, then a mass balance for the pollutant at any location in the discharge plume can be written as:

$$(Q_a * C_a) + (Q_e * C_e) = (Q_a + Q_e)C_p$$

$$\left(\frac{Q_a}{(Q_a + Q_e)} * C_a\right) + \left(\frac{Q_e}{(Q_a + Q_e)} * C_e\right) = C_p$$

If X = $\frac{Q_e}{(Q_a + Q_e)}$ = fraction of effluent in the plume

Then, 1-X = $\frac{Q_a}{(Q_a + Q_e)}$ = fraction of ambient water entrained in the plume

$$(1 - X)\ C_a + X\ C_e = C_p$$

$$X = \frac{(C_p - C_a)}{(C_e - C_a)}$$

And by definition,

$$DF = \frac{1}{X} = \frac{(C_e - C_a)}{(C_p - C_a)} \tag{2}$$

If the background concentration (C_a) is zero,

$$DF = \frac{1}{X} = \frac{C_e}{C_p}$$

A DF calculated using this equation is an empirical result for the particular sampling point where the C_p value is measured.

At the source, $C_a = 0$ and $C_p = C_e$, therefore DF = 1. Most mixing zone models use this definition.

An initial dilution model generates dilution factors using outfall, effluent, and receiving water characteristics supplied to it. Each DF that prints out is for a particular calculated distance as the model iterates along the plume trajectory away from the outfall. Depending on the model used, the DF (and C_p) may be calculated simply using the volume fraction equation (Equation 1), or the C_p may be calculated as an actual, effective diluted concentration (depending on whether the model accepts C_e and C_a as inputs).

Rearranging equation (2) gives:

$$C_p = C_e\left(\frac{1}{DF}\right) + C_a\left(1 - \left(\frac{1}{DF}\right)\right) \tag{3}$$

Again depending on the model used, the printout may occur repeatedly reflecting the model's iterative process along the plume trajectory or it may occur only upon completion of initial dilution. Whatever the capability of the model, it is imperative that its generated C_p can be validated, *i.e.*, compared to measured C_p at the same distance from the outfall to establish how well the model is simulating the plume. A dye tracer is generally better for this task because dye can be measured *in situ* with a fluorometer.

4. Prorating Dilution Factors

Prorating of the dilution factor is necessary when two or more dischargers share the same outfall. The general approach is to assign a flow-proportioned dilution factor for each of the dischargers sharing the outfall. A second approach is for stormwater flows where an area proportioned dilution factor may be used for each of the dischargers sharing a common outfall. A reasonable potential analysis should be conducted based on the prorated dilution factor for each discharge.

Several factors need to be considered here:

a. When a specific pollutant exists in only one of the dischargers and the pollutant concentration is measured at the particular site (as opposed to at the combined outfall), additional dilution may be granted based on flow from other sites.

b. If for a specific site only the contact-stormwater flow is treated while the stormwater flow from rest of the site is "clean" and the two flows are combined before leaving the site, and the pollutant is measured at the end of the treatment system, then additional dilution may be allowed in excess of the area-proportioned dilution to the extent that then "clean" stormwater dilutes the treated stormwater.

5. Centerline and Flux-Average Dilution Factor

Ecology requires that for unidirectional ambient flows, the dilution factor be based on plume centerline (peak) concentration at the edge of a mixing zone. The presumption is that the plume centerline would have the highest concentration (see Figure C-1) and would result in the lowest dilution factor across the plume cross-section.

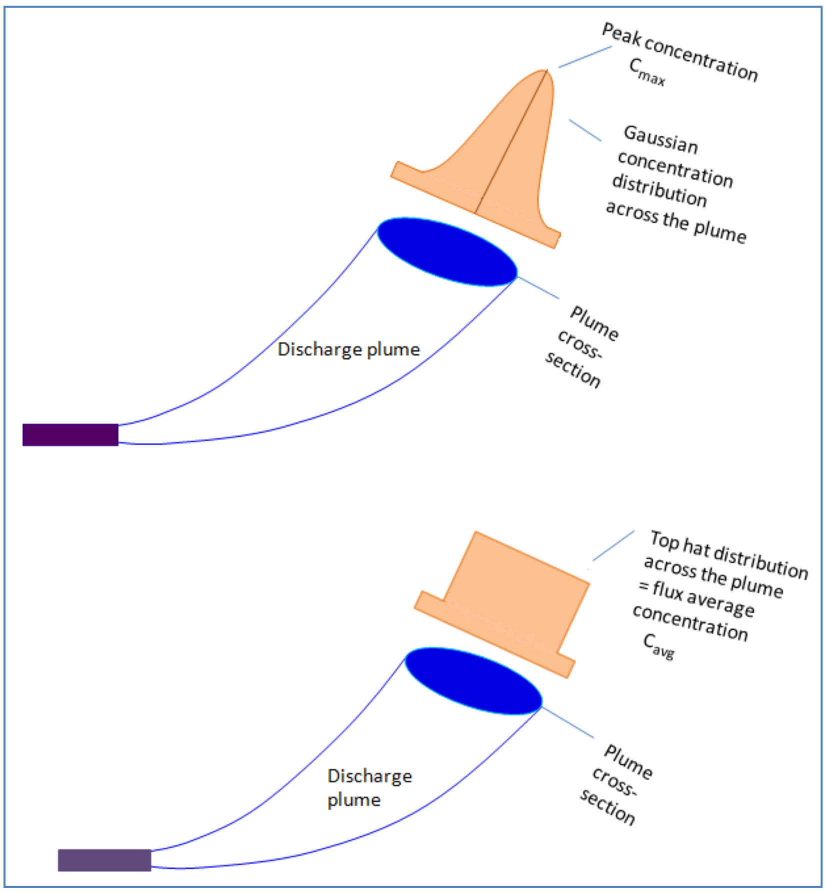

Figure C-1. Pollutant concentration distribution across a discharge plume

For multidirectional flow conditions, such as tidal areas, the dilution factor should be based upon a flux-average plume concentration at the edge of a mixing zone. However, tidally influenced rivers where flows do not reverse and thus a unidirectional flow is clearly established, a centerline dilution factor may be warranted.

Plume velocities in a cross section of each building block (perpendicular to the path of the trajectory) resemble a bell-shaped curve. Concentrations, on the other hand, do not resemble a bell-shaped curve (*i.e.*, peak concentrations do not occur at the same location as the center-of-mass). Therefore, an average concentration involves weighting the concentration distribution by the velocity distribution. This average may be referred to as either a “top hat” or “flux-average”, depending upon how it is formulated in a particular model. It is the value to be multiplied by the total plume volume flux to get total mass flux, which is passed on to the farfield algorithm.

The peak to mean ratio is simply the ratio of the centerline to the average concentration. It is obtained from a flux integral. Starting with the relationship for average concentration (C_{avg}):

$$C_{avg} = \frac{\int_A CvdA}{\int_A vdA}$$

where C and v are the instantaneous concentration and velocity in the plume element dA.

The peak to mean ratio is then defined as Cmax/Cavg,

$$\frac{C_{\max}}{C_{avg}} = \frac{C_{\max}\int_A vdA}{\int_A CvdA}$$

where C_{max} is the centerline concentration.

Mixing zone models predict either a centerline or average (top hat) or both concentration of pollutant across a plume cross-section. Then, based on an assumed distribution (e.g., 3/2 power profile or Gaussian), the peak to mean ratio is estimated which is then used to estimate the missing concentration (either centerline of flux-average). Other factors influencing the peak to mean ratio may include whether adjacent plumes have merged, the plume geometry, and location on the plume trajectory.

Review each model output and documentation to establish the peak to mean ratio.

- In RiverPlume6, the output includes both a flux-average dilution factor as well as a dilution factor at point of interest. The latter being a centerline dilution factor if centerline coordinates are specified. It should be noted that the downstream flow entered in RiverPlume6 is sum of both the critical ambient flow as well as effluent flow. This being the case, the definition of dilution factor discussed above holds true for RiverPlume6 as well.

- In CORMIX output, for submerged plume or jet regions a minimum centerline dilution (Sc) is predicted, while for buoyant spreading regions a flux-average (or bulk) dilution (Sf) is predicted. If one or the other needs to be estimated, the following relationship can be used: Sf = 1.7Sc or 1.3Sc depending on whether it is single port (submerged/surface discharge), or a multiport (line discharge).

- In the latest version of Visual Plumes the peak to mean ratio used is 2 for unmerged plumes and 1.5 for merged plumes past the zone of flow establishment (generally a distance greater than 6 times port diameter) where the top-hat concentration distribution of the effluent transitions to a Gaussian distribution.

- For farfield predictions within Visual Plumes utilizing Brook's (1960) equations (see section on "Choosing a Farfield Model") following nearfield predictions by UM3 model (see section on "Understanding Initial Dilution Theory") remember the following:

  The initial condition in Brooks farfield prediction is a constant concentration across the wastefield equivalent to the average concentration predicted at the end of the nearfield mixing regime. Beyond this point, the wastefield spreads, due to lateral dispersion, and ultimately approaches Gaussian distribution. For wide wastefields, it takes sometime before the centerline concentration begins to decline, which is why you often see concentration fairly constant for some distance from the source region. Flux average concentration predictions from Visual plumes can be used to predict centerline concentration predictions (C_{max}) using Brooks error function (Brooks 1960).

6. Selecting a Reasonable Worst-Case Scenario

Aquatic life-based analyses involve the concept of determining reasonable worst-case values for various parameters because the durations established for these water quality criteria vary for both acute and chronic toxicities. For acute toxicity the durations are instantaneous (e.g. silver), one-hour (e.g. ammonia), and twenty-four hours (e.g. polychlorinated biphenyls or PCB). For chronic toxicity the durations are twenty-four hours (e.g. PCB) and four-days (e.g. zinc). There are two types of human health-based water quality criteria: Those based on non-cancer effects and those based on cancer effects. The same concept of reasonable worst-case applies in non-cancer analyses as applies in aquatic life-based analyses. The concept of average values applies to carcinogenic human health-based analyses because the duration established for these criteria is the average life span of a person.

The term *reasonable worst-case* refers to a selected value for a specific effluent or receiving water parameter (e.g.**,** reasonable worst-case current). *Critical condition* refers to a scenario involving reasonable worst-case parameters, which has been set up to run in a mixing zone model (e.g., critical condition scenario to determine mixing at the chronic boundary). Steady-state mixing zone models are usually applied using a combination of parameters (e.g., effluent flow, current speed, depth, density) packaged to simulate either a critical or an average condition. It is understood that each critical condition (by itself) has a low probability of occurrence. Discharges to tidally-influenced rivers where a saltwater wedge is present may warrant special consideration of critical conditions which are known to occur simultaneously (e.g., during low tides, the predominance of freshwater may always create a well-mixed profile; while during high tides a stratified profile may always exist).

A mixing zone analysis should generally include a sensitivity analysis, particularly when the model input parameters have a high degree of uncertainty or on impact when dilution is needed to be evaluated for a particular model input. A sensitivity analysis is a series of scenarios organized such that only one reasonable worst-case parameter in each scenario is changed while all others are held constant in a logical progression.

Those reasonable worst-case and average parameters that are required input to a model are discussed in Section 6. Section 7 addresses other factors which must be considered before arriving at the correct dilution factors for the acute and chronic boundaries: The Standards require that mixing zones not occupy more than a certain percentage of the channel width and that the effluent flow rate not utilize more than a certain percentage of the available receiving water flow rate in the process of dilution. So actually, the dilution factor to use when determining whether the effluent has a reasonable potential to exceed water quality criteria must be the lowest one of three that can be generated for both the acute and chronic boundaries.

6.1 Critical Effluent Flow

It is important to consider critical effluent flows so that the lowest dilution that is protective of water quality can be estimated. These conditions are shown in Table C-1.

Table C-1. Point source steady-state flows for mixing zone analysis

Source	Mixing Zone	Aquatic life or Human health (Carcinogens, Non-Carcinogens)	Point Source flow used for steady state mixing zone analysis during critical period
Municipal wastewater Treatment Plant discharge	Acute	Aquatic life	Consider data from last 3 years a. If monthly Q_{max} < 0.85 Q_{design} (dry weather) use daily Q_{max} b. If monthly $Q_{max} \geq 0.85$ Q_{design} (dry weather) use peaking factor x Q_{design} where peaking factor = (daily Q_{max})/(monthly Q_{max})
	Chronic	Aquatic life and Non-Carcinogen	Consider data from last 3 years a. If monthly Q_{max} < 0.85 Q_{design} (dry weather) use monthly Q_{max} b. If monthly $Q_{max} \geq 0.85$ Q_{design} (dry weather) use Q_{design}
		Carcinogen	Use annual Q_{avg} either design or projected over the life of the permit
Industrial wastewater discharge	Acute	Aquatic life	Use daily Q_{max} for past three years or projected over the life of the permit during critical period
	Chronic	Aquatic life and Non-Carcinogen	Use monthly Q_{max} for past three years or projected over the life of the permit during critical period
		Carcinogens	Use annual Q_{avg} either design or projected over the life of the permit
Storm water (Western Washington)	Acute	Aquatic life	Use 1-hour Q_{max} with recurrence interval of once every three years
	Chronic	Aquatic life	Use 96-hour Q_{max} with recurrence interval of once every three years

6.1.1 Stormwater Flow Rate

For analyses at the acute boundary, at the acute boundary, the stormwater flow rate to use in Western Washington is the peak one-hour flow rate with a recurrence interval of once every 3 years as obtained from 50-plus years of precipitation data using Western Washington Hydrology Model (WWHM) (https://ecology.wa.gov/Regulations-Permits/Guidance-technical-assistance/Stormwater-permittee-guidance-resources/Stormwater-manuals/Western-Washington-Hydrology-Model) or other approved models.

For analyses at the chronic boundary, the stormwater flow rate to use in western Washington is the peak 96-hour flow rate with a recurrence interval of once every three years as generated from about 50-plus years of precipitation data using WWHM or other approved models. A procedure to estimate these flows using the WWHM model is included in Attachment C.1.

Other stormwater flow rates may be estimated that have a duration and recurrence interval associated with specific water quality standards. For example, for an "instantaneous not to be exceeded at anytime criteria" a 99th percentile of maximum annual hourly average flow rate data over the period of record should be used, and for a "24 hour average not to be exceeded criteria" a 99th percentile of maximum annual daily average flow rate data over the period of record should be used.

Because most human health-based criteria are based on lifetime exposures, direct comparisons of receiving water criteria with pollutant concentrations in intermittent stormwater discharges are not appropriate. This and the high variation in stormwater pollutant concentrations and discharge volumes, both between storms and during a single storm, make the application of human health criteria to stormwater particularly problematic. Based on the authority of 40 CFR 122.44(k)(3), in the Industrial Stormwater General Permit, Ecology requires the implementation of best management practices to control or abate pollutants because it is infeasible to derive appropriate numeric effluent limits for the human health criteria.

6.1.2 Intermittent Effluent Flow

Steady-state (averaged) effluent flow is a commonly accepted approximation of inherent variability – but only for continuous discharges. When effluent discharge is intermittent, the reasonable worst-case flow rate to use is the maximum that can occur – whether through pumps or gravity flow.

The dilution factor generated using the maximum flow rate may then be adjusted upward by a ratio of the maximum flow to the appropriate time-averaged flow for the criterion being assessed. For aquatic life criteria, acute dilution factors are typically assessed using the maximum one-hour average flow. Chronic dilution factors are typically assessed using the maximum 4-day average flow. However, the appropriate flow averaging period varies per pollutant. The modeler should reference the water quality standards (see Table C-2) for the appropriate assessment duration. For human health carcinogens the appropriate effluent flow is the highest annual average flow. For human health non-carcinogens the appropriate effluent flow is the highest monthly average flow (*Permit Writer's Manual*, Chapter 7).

Example:

Effluent flow rate = 850 gpm, for 25 minutes 3 times a day.

- The equivalent one-hour flow rate is 354 gpm (=850 *25/60).
- The equivalent 24-hour flow would be 44 gpm (= 850 *25*3/24*60).
- The equivalent 4-day average flow = 24-hour flow rate (since the flow pattern is the same every day)
- The equivalent monthly average flow = 24-hour flow rate (since the flow pattern is the same every day)

Use the peak flow rate (850 gpm) and respective ambient critical conditions to estimate dilution factors at the edge of acute (DF_acute(peak)) and chronic mixing zones (DF_chronic(peak)) using an appropriate model. Then,

- Actual dilution factor at the edge of acute zone, DF_acute = (850/354)* DF_acute(peak)
- Actual dilution factor at the edge of chronic zone, DF_chronic = (850/44)* DF_chronic(peak)

Table C-2. Aquatic Life Criteria Durations

Substance	Freshwater		Marine Water	
	Acute	Chronic	Acute	Chronic
Aldrin/Dieldrin	instantaneous	24-hour	instantaneous	24-hour
Ammonia (un-ionized NH3)	1-hour	4-day	1-hour	4-day
Arsenic	1-hour	4-day	1-hour	4-day
Cadmium	1-hour	4-day	1-hour	4-day
Chlordane	instantaneous	24-hour	instantaneous	24-hour
Chloride (Dissolved)	1-hour	4-day	-	-
Chlorine (Total Residual)	1-hour	4-day	1-hour	4-day
Chlorpyrifos	1-hour	4-day	1-hour	4-day
Chromium (Hex)	1-hour	4-day	1-hour	4-day
Chromium (Tri)	1-hour	4-day	-	-
Copper	1-hour	4-day	1-hour	4-day
Cyanide	1-hour	4-day	1-hour	4-day
DDT (and metabolites)	instantaneous	24-hour	instantaneous	24-hour
Dieldrin/Aldrin	instantaneous	24-hour	instantaneous	24-hour
Endosulfan	instantaneous	24-hour	instantaneous	24-hour
Endrin	instantaneous	24-hour	instantaneous	24-hour
Heptachlor	instantaneous	24-hour	instantaneous	24-hour
Hexachlorocyclohexane (Lindane)	instantaneous	24-hour	instantaneous	-
Lead	1-hour	4-day	1-hour	4-day
Mercury	1-hour	4-day	1-hour	4-day
Nickel	1-hour	4-day	1-hour	4-day
Parathion	1-hour	4-day	-	-
Pentachlorophenol (PCP)	1-hour	4-day	1-hour	4-day
Polychlorinated Biphenyls (PCBs)	24-hour	24-hour	24-hour	24-hour
Selenium	1-hour	4-day	1-hour	4-day
Silver	instantaneous	-	instantaneous	-
Toxaphene	1-hour	4-day	1-hour	4-day
Zinc	1-hour	4-day	1-hour	4-day

Source: WAC 173-201A-240

6.1.3 Combined Sewer Overflow Discharges

Combined Sewer Overflows (CSOs) are intermittent discharges and can occur through either a 1) untreated CSO outfall or a 2) outfall associated with a satellite CSO treatment plant (i.e. treated CSO discharge).

1) Untreated CSO outfall – Once it is controlled to meet the performance standard in WAC 173-245-020(22), the controlled outfall is allowed a once per year exemption to the numeric mixing zone criteria per WAC 173-201A-400(11), provided that there is no potential to cause a loss of sensitive habitat, interfere with the existing or characteristic uses of the waterbody, result in damage to the ecosystem or adversely affect public health [WAC 173-201A-400(4)]. Ecology considers controlled untreated CSO outfalls to have sufficient dilution to be in compliance with the aquatic life and human health criteria, assuming the above conditions are met.

2) Satellite CSO treatment plant discharges are considered treated discharges and typically discharge more frequently than once per year. They do not qualify for the exemption to the numeric mixing zone criteria. Because these are treatment plants and apply AKART, CSO satellite treatment plants have mixing zones and associated dilution factors. CSO treatment plant discharge events are relatively infrequent with durations typically shorter than chronic aquatic life criteria durations. The modeler should reference the water quality standards (see Table C-2) for the appropriate assessment duration. Take into consideration any increases in plant flows expected during the life of the permit.

Example:

Satellite CSO Treatment Plant: discharge happens approximately 10 times per year. Each discharge event lasts between 1 and 96 hours.

- The instantaneous flow rate is the maximum instantaneous flow rate recorded (or projected over the next permit term)
- The one-hour flow rate is the maximum one-hour flow rate recorded (or projected over the next permit term).
- Equivalent 24-hour flow: For each day with effluent discharge, calculate total volume discharged. Use the highest equivalent 24-hour flow.
- Equivalent four-day flow = Highest total 4-day discharge event volume divided by 4 days. If the CSO discharge is less than four days, use highest total event volume divided by 4 days.

Satellite CSO treatment plant discharges are highly intermittent and highly variable in discharge volumes, durations, and pollutant concentrations, both between storms and during a single storm event. Therefore, direct comparisons of human health receiving water criteria with pollutant concentrations is not appropriate. Deriving numeric effluent limits for human health criteria is infeasible. Based on the authority of 40 CFR 122.44(k)(3), Ecology requires the implementation of best management practices (BMPs) to control or abate human health pollutants from satellite CSO treatment plants.

6.2 Critical Ambient Conditions

Table C-3 includes guidelines for establishing critical ambient characteristics for use in calculating dilution factors

Table C-3. Critical ambient flow, velocity, and depth for mixing zone analysis

Source	Mixing Zone	Aquatic life or Human health (Carcinogens, Non-Carcinogens)	Flow, velocity and depth for mixing zone analysis under critical conditions
Freshwater	Acute	Aquatic life	Use flow, velocity, depth and width under 7Q10 low flow conditions
	Chronic	Aquatic life	
		Carcinogens	Use flow, velocity, depth and width under harmonic mean flow
		Non-carcinogens	Use flow, velocity, depth and width under 30Q5 low flow conditions. If 30Q5 not available, use 7Q10 flow conditions.
Salt water and tidally influenced freshwater	Acute	Aquatic life	Critical 10th and 90th percentile current velocities are derived from a cumulative frequency distribution analysis over one neap and spring tide cycle. The velocity with lowest dilution is the critical velocity. Critical ambient density profile is defined as the density profile that results in the lowest mixing. The diffuser depth is defined as the depth at MLLW (marine waters) or at MLLW during a 7Q10 low flow period (tidally influenced freshwater regions).
	Chronic	Aquatic life	Critical receiving water current velocity is defined as the 50th percentile current velocity derived from a cumulative frequency distribution analysis over at least one tidal cycle. The critical ambient density profile is defined as the density profile that results in the lowest mixing. The diffuser depth is defined as the depth at MLLW. For marine waters, the diffuser depth is defined at MLLW. For tidally influenced freshwater regions, the diffuser depth is at MLLW during low 7Q10 (aquatic life), low 30Q5 (non-carcinogens) or harmonic mean flows (carcinogens).
		Carcinogens	
		Non-carcinogens	

6.2.1 Tidally-Influenced Waterbodies

For analyses at the acute boundary in tidally-influenced water, the velocity to use is the critical 10th and the 90th percentile velocities derived from a cumulative frequency distribution analysis (see Attachment C-2). The distribution analysis should be produced from a data set consisting of periodic readings taken by an instrument deployed over a neap and spring tide cycle. In the absence of a comprehensive field data set, a sensitivity analysis should be run using a wide range of possible velocities which could reasonably occur for any 1-hour duration. The velocity which produces the lowest dilution should be considered the critical velocity.

For analyses at the chronic boundary in tidally-influenced water, the critical velocity is defined as the 50th percentile current velocity derived from a cumulative frequency distribution analysis. In the absence of a comprehensive field data set, a sensitivity analysis should be run using a wide range of velocities, any of which could reasonably occur as the average velocity for any 4-day duration. The velocity which produces the lowest dilution should be considered the critical velocity.

6.2.2 Density Profile

The density of seawater depends on temperature, salinity, and pressure (which increases with water depth). The density is sometimes denoted in terms of sigma-t. The density of freshwater at standard temperature and pressure is 1 gm/cm^3. A water sample with a density of 1.027 g/cm^3 has a sigma-t value of 27. The density profile to use in aquatic life-based analyses is the one that results in the least mixing. Generally, this is either the minimum or maximum stratification, defined as follows: "Minimums" are characterized by profiles that extend to the same depth as the outfall with the smallest differential between sigma-t values at the bottom and top of the effluent plume. "Maximums" are characterized by profiles that extend to the same depth as the outfall with the largest differential between sigma-t values at the bottom and the plume trapping depth. Some profiles which are profoundly nonlinear warrant more thoughtful consideration. In Puget Sound, changes in density correlate most closely to changes in season (Glenn and Giglio, 1997). Minimum stratifications frequently occur in October, while maximum stratifications frequently occur from May 1-July 15. There is little or no correlation between changes in stages of tide and changes in profiles.

The density profile to use in human health-based analyses is the one that results in average mixing. This is determined as follows: (1) Generate the dilution factors for the two profiles (minimum and maximum), (2) calculate the reciprocal of the dilution factors to convert them to effluent concentrations, (3) calculate the average of the reciprocal dilution factors (average effluent concentration), and (4) calculate the reciprocal of the average effluent concentration and use that as the harmonic mean dilution factor.

7. Minimize Mixing Zones

WAC 173-201A-400(6) requires that the size of the mixing zone be minimized. To accomplish this, the following restrictions must be included in establishing the dilution factor and, in certain circumstances, you may recalculate it at a boundary smaller than the maximum allowable mixing zone size.

7.1 Maximum Downstream Distance Allowed

Maximum sizes of mixing zones for rivers, estuaries and open ocean are defined in WAC 173-201A-400(7). In rivers and streams the maximum mixing zone boundary is 300 feet downstream plus depth of diffuser at 7Q10 flow. In estuaries (as defined in WAC 173-201A-400(7)(b)(ii), see Attachment C-3) the maximum mixing zone boundary is 200 feet plus depth of diffuser at MLLW in any horizontal direction. In oceanic waters the maximum mixing zone boundary is 300 feet plus depth of diffuser at MLLW in any horizontal direction. A zone of acute water quality criteria (WAC 173-201A-240) exceedance of 10% of the mixing zone size described above is allowed as described in WAC 173-201A-400(8).

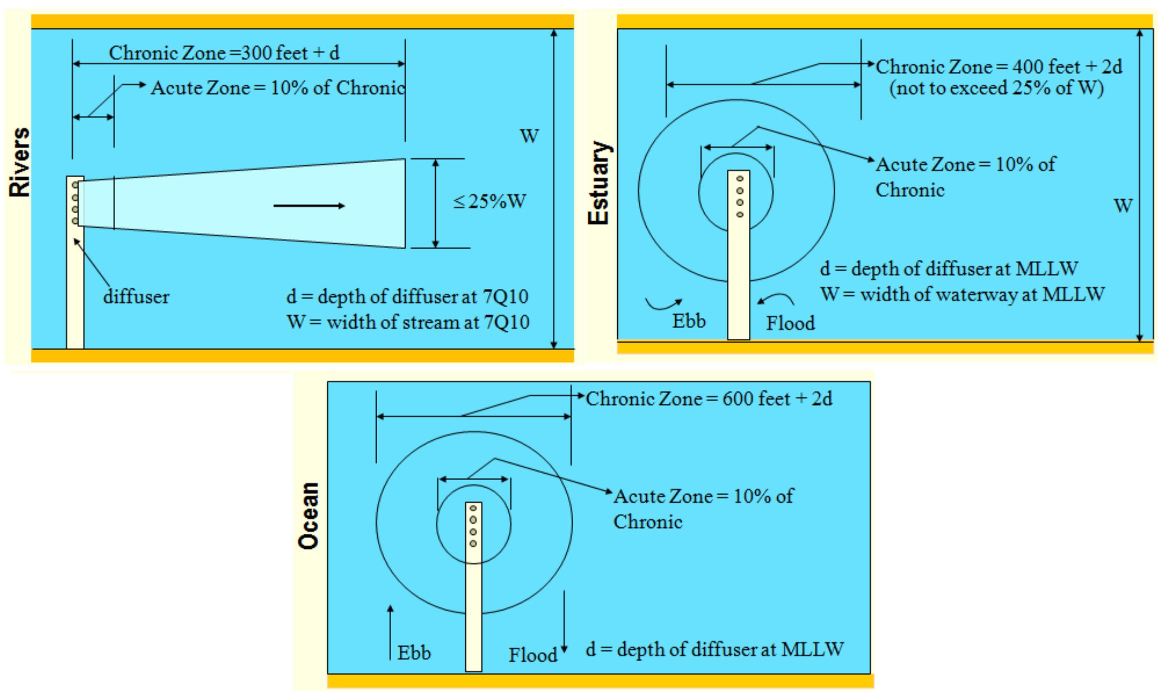

Figure C-2. Mixing zone size restrictions in rivers, estuaries and open ocean

In estuaries that have river like flow characteristics, the maximum size restrictions for a stream may be applied as per WAC173-201A-400(7)(a). Generally, this should be taken to mean that the flow in the tidally influenced river never reverses.

7.2 Maximum Waterbody Width Restrictions

Maximum width (Wmax) of a water body that can be “occupied” by both the acute and chronic mixing zones cannot be more than twenty-five percent of the width (W) of the water body under critical conditions. The Channel width must be determined during a 7Q10 (in freshwater), MLLW (in estuaries), or combination thereof (tidally-influenced riverine waters).

$$W_{max} \leq 0.25\ W$$

7.3 Maximum Ambient Flow Restrictions

Maximum flow rate in non-tidally reversing rivers and streams that can be “utilized” by a chronic mixing zone is 25% and by an acute mixing zone is 2.5%. Formulation of this dilution factor for an entire receiving water involves solving the volume fraction equation:

$$DF_{\max\_chronic} = \frac{(0.25 * Q_{amb} + Q_e)}{Q_e}$$

$$DF_{\max\_acute} = \frac{(0.025 * Q_{amb} + Q_e)}{Q_e}$$

Where

$DF_{max\_}$ =maximum dilution factor allowed under chronic and acute conditions

Q_{amb} is the critical flow rate of a receiving (ambient) water; and

Q_e is the critical flow rate of effluent.

7.4 Applying Distance, Width, and Flow Restrictions to Model-Predicted Dilution Factor

Table C-4 shows four cases where the maximum dilution factor allowed based on flow restriction (DF_{max}) and the maximum width allowed (W_{max}) is compared with model-predicted dilution factor (DF) and model-predicted plume width (W).

Table C-4. Evaluating different combinations of maximum allowable flow based dilution and maximum width restrictions

Case	regulatory max	logical	estimated	DF used	subsequent analysis
case 1:	DF_{max}	>	DF	DF	none
	W_{max}	>	W		
case 2:	DF_{max}	<	DF	DF_{max}	none
	W_{max}	>	W		
case 3:	DF_{max}	>	DF	cannot use DF	Find new DF by reducing mixing zone distance until $W < W_{max}$
	W_{max}	<	W		
case 4:	DF_{max}	<	DF	cannot use DF_{max}	Find new DF by reducing mixing zone distance until $W < W_{max}$
	W_{max}	<	W		

W_{max} maximum regulatory width allowed at the edge of mixing zone
W width of plume estimated at edge of mixing zone
DF_{max} maximum regulatory dilution allowed at edge of mixing zone
DF dilution factor estimated at edge of mixing zone

When the mixing zone distance needs to be reduced because the applicable width restriction cannot be met, then further evaluation is necessary. Dilution factors and plume widths predicted by the model for different distances from outfall are compared with both maximum allowable flow-based dilution factor as well as applicable width restrictions. The distance at which these restrictions are met is then the new mixing zone size.

Figure C-3 shows how dilution factors should be minimized based on size, width and flow restrictions included in WAC 173-201A-400. This figure shows the maximum dilution factor allowed based on flow restriction as 30. The maximum distance of the mixing zone allowed under regulation is 300 ft. The maximum width is restricted to 13 ft. In order to find out which of these is most restrictive, we begin by plotting plume width versus distance as well as dilution factor versus distance as predicted by a mixing zone model. The plot shows that the plume width restrictions govern. Locate on the plume width curve where it is 13 ft. Extend from this point to the x-axis to locate the maximum size of the mixing zone. Also, for the 13 ft width of plume locate the dilution factor (DF = 18.7) on the DF curve.

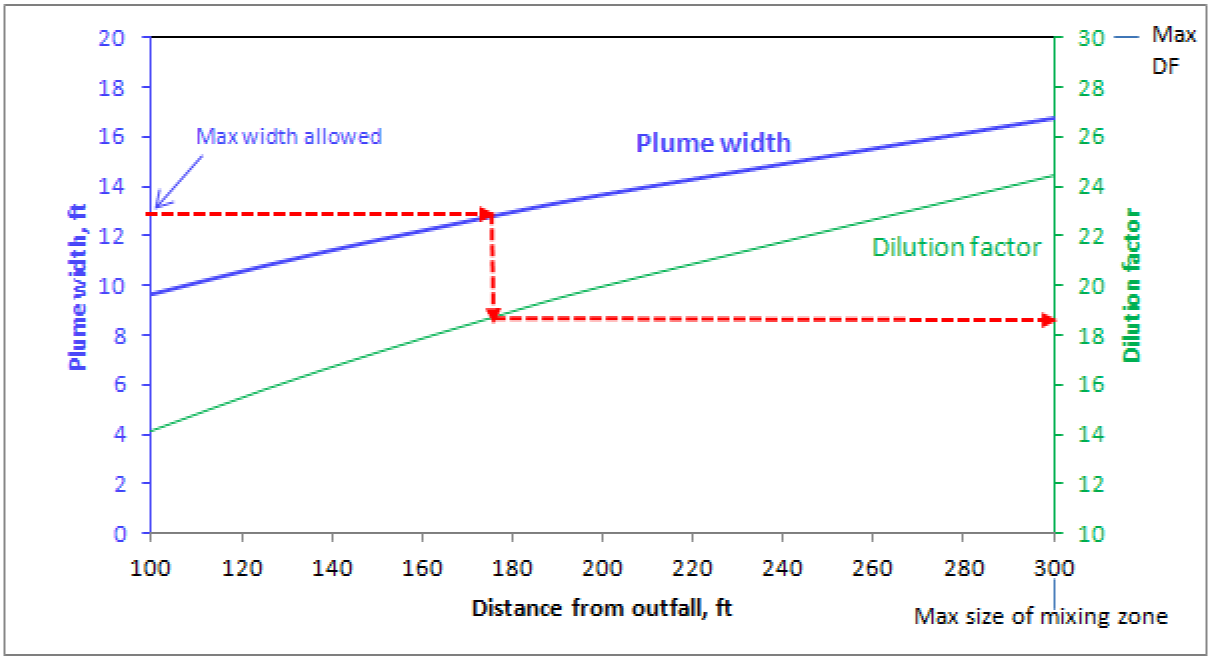

Figure C-3. Example of minimizing dilution factor

8. Choosing an Initial Dilution Model

The physical mixing of effluent discharges in ambient receiving waters depends on many factors as discussed below. The most important consideration for model selection is "does the model simulate the physical mixing process likely to occur within the regulatory mixing zone?" Therefore a discussion of factors affecting physical mixing process is presented here:

i. Discharge characteristics, whether it is a "pure jet" (buoyancy is negligible) and exit velocity dominates the nearfield mixing; or it is a "pure plume" (exit velocity is negligible); or whether it is a combination of the two.

ii. Ambient characteristics, whether the receiving water is stratified or not. This would govern whether discharge would be trapped below the surface or not.

iii. Whether nearfield or farfield mixing processes dominate at the location where dilution factor is being sought.

   a. Nearfield mixing is dominated by source geometry and effluent characteristics discussed above, farfield mixing processes are dominated by ambient receiving water characteristics and is driven by dispersion (or passive diffusion).

   b. The dispersion characteristics of ambient receiving waters depends on whether it is predominantly uni-directional flow, or tidally reversing flow as in an estuary or in on open ocean; ambient current speed; channel morphology; and presence of eddies

iv. Whether the effluent is rapidly and uniformly vertically mixed in the receiving water, e.g., in a shallow river. In this case farfield dynamics (lateral mixing) dominate. The lateral dispersion depends upon the type of receiving water as discussed above.

v. Boundary effects such as attachment of discharge plume to bank, bottom, surface, or a stratified terminal layer. This tends to reduce dilution. Bottom attachments can be "wake" attachment that occurs when strong ambient currents bends the plume over, or "coanda" attachment that occurs when entrainment demand of the plume, caused by a low pressure region between the jet and the bottom, bends the plume over. Bottom attachments can have potential benthic impacts.

vi. Bouyant upstream intrusions caused by upstream density currents and weak ambient crossflow current

vii. Configuration of diffuser ports and nozzles also play an important role in dictating the magnitude of initial mixing.

viii. Re-entrainment of discharge in the nearfield due to surface or bottom interactions of plumes and local recirculation patterns can reduces dilution since it increases pollutant concentrations within the plume.

In choosing a mixing zone model it is important to select models that appropriately captures/simulates the ambient/discharge interaction at a given waterbody location. A prior validation of the model (e.g. with a dye study or from literature review) is helpful in justifying the selection of the model.

It is important to also consider whether to use simple models that underestimate dilution and lead to overprotective permit limits, or to use a complex model that may provide a dilution factor that is more accurate, but comes at a cost associated with purchasing/maintaining the complex model; and the cost may not be necessarily recouped with a less restrictive permit limit.

8.1 Understanding Initial Dilution Theory

The general theory behind wastefield formation is easily understood. Visualize wastewater discharged horizontally as a jet from a single round port (as in Figure C-1) or a series of jets from ports spaced at equal distances along a diffuser. Two forces shape the immediate nature of the plume. The first is the exit velocity of the discharge which would move the plume along the exit angle of the discharge. If the pipe is horizontal, as is the case in Figure C-4, the discharge plume will be pushed horizontally. The second force is due to density differences between the discharge and the ambient. If the wastewater has a lower density than the surrounding water, then the resulting buoyancy force deflects the jet(s) upward forming plumes which are swept downstream by the current.

The plume(s) entrain ambient water as they rise, causing them to be diluted and decreasing the density difference between them and the ambient. If the ambient water is stratified, then its density at the depth of the ports is greater than near the surface. The greater density ambient water is entrained initially, and the rising, expanding plumes can reach a level where their density is the same as the surrounding water (i.e., neutral buoyancy). This is the trapping depth.

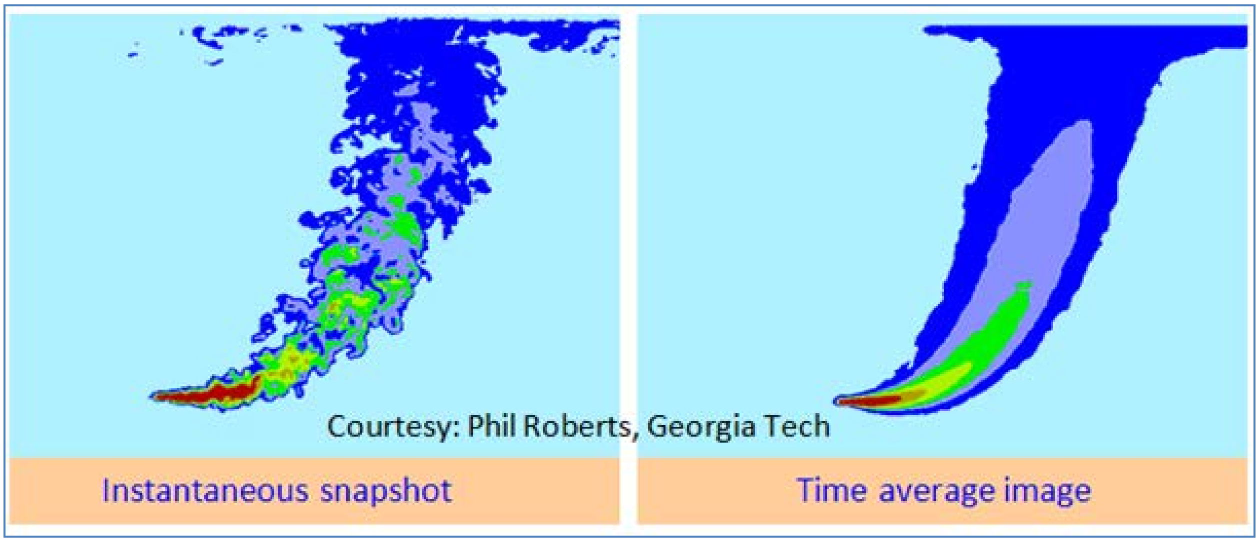

Figure C-4. Plume schematic showing instantaneous photo with turbulent mixing (left), time-averaged plume with distinctive concentration contours (right).

8.2 Choosing a Nearfield Model

Nearfield models predict plume characteristics, under steady state, in the vicinity of the discharge outfall, primarily governed by discharge exit velocity and buoyancy. Under steady-state conditions, the plume shape is stationary. If we follow an element of the plume exiting from a port and moving from the discharge point to the maximum rise and beyond, driven by inertia and other forces, the progression of the element produces the persistent plume shape under steady-state discharge and ambient conditions. Several nearfield models, both empirical as well as theoretical models are discussed below.

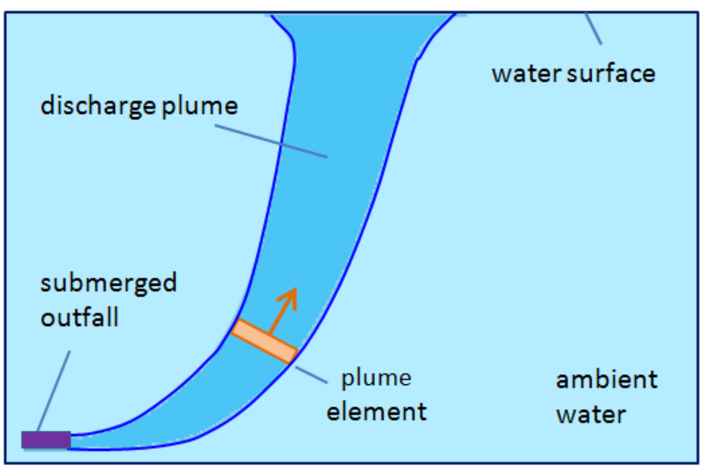

Figure C-5. Plume element exiting from discharge pipe and rising in the ambient water under jet momentum and buoyant forces.

8.2.1 Simplistic Screening Level Minimum Dilution Estimation

A minimum estimation of initial dilution can be made assuming that the exit velocity is zero and the discharge is neutrally buoyant and that the discharge pipe is flowing full. This simplest approach is modeled using the “Jet momentum Equation” as described in EPA (1991). The dilution is underestimated and is only valid up to a distance of 2 to 3 times the water depth. That is, it is an estimation of nearfield dilution and should be used as a screening tool.

$$S = 0.3\frac{x}{d}$$

where S = flux average dilution
x = distance from outlet
d = diameter of outlet

8.2.2 Maximum Dilution Factor in Rivers

As discussed earlier, the maximum dilution factor in rivers that can be allowed under Washington State law is given by the following zero-dimensional model. This approach is also referenced in EPA (1985).

$$DF_{\max\_acute} = \frac{(0.025 * Q_{amb} + Q_e)}{Q_e}$$

where
DFmax_acute = maximum dilution factor allowed at the edge of mixing zone
Q_{amb} = 7Q10 flow
Q_e = critical effluent flow

8.2.3 RiverPlume6

This is a one -dimensional model based on the method described in Fischer et al. (1979). Thus, it is strictly a farfield model and a detailed description is included under the section *Understanding Farfield Theory*. However, if the discharge plume is completely vertically mixed within a short distance, this model may be used for unidirectional ambient water bodies to predict dilution in both nearfield and farfield regions. The applicability of the model and inherent assumptions are discussed later.

8.2.4 CORMIX

This is an EPA-supported mixing zone model and decision support system for estimating dilution resulting from continuous point source discharges. The system emphasizes the role of boundary interaction to predict steady-state mixing behavior and plume geometry. CORMIX stands for CORnell MIXing zone models. The package consists of CORMIX1, CORMIX2, and CORMIX3 for the analysis of submerged single port discharges, submerged diffusers, and surface discharges, respectively.

CORMIX consists of a large number of mixing zone model equations for both near field and far field conditions. The near and far field models within CORMIX are not distinctly separated as in other models like UM3 discussed below. It contains an analytical scheme that classifies any given discharge/ambient situation into one of several categories with distinct hydrodynamic features. Based upon the site-specific conditions provided by the model user, the "expert system" in CORMIX automatically chooses and applies the mixing zone model equation appropriate for the system under study. The strengths of this model are its ability to acknowledge the effects of boundary constraints and gravitational collapse. Plume centerline dilution and concentrations can be easily converted to bulk/flux-average dilution and concentrations as discussed earlier. CORMIX gives full 3-dimensional plume trajectory, and plume travel time outputs. Lateral concentrations can be calculated with the presumed Gaussian plume profiles.

Detailed information on CORMIX including information on license and link to user manual is available at http://www.cormix.info/index.php.

The CORMIX methodology contains systems to model single port (CORMIX1), multiport (CORMIX2) and surface (CORMIX3) discharges. CORMIX1 also has the capability to model above surface (free falling) discharges.

a. **CORMIX1** deals with submerged single port discharges (Figure C-6) into flowing unstratified or stratified water environments, such as rivers, lakes, estuaries, and coastal waters. It includes the limiting cases of non-buoyant and negatively buoyant discharges and of stagnant ambient conditions. It also deals with tidal reversing flow conditions and arbitrary ambient current and density profiles with CorJet, which is an advanced integral model tool for detailed nearfield analysis. CORMIX1 also deals with discharges above the water surface (Figure C-6) where the free falling discharge plunges into ambient water and eventually rises up, creating a discharge plume due to buoyancy. Notice that the defined outline of discharge plumes in Figure C-6 represents a steady-state condition.

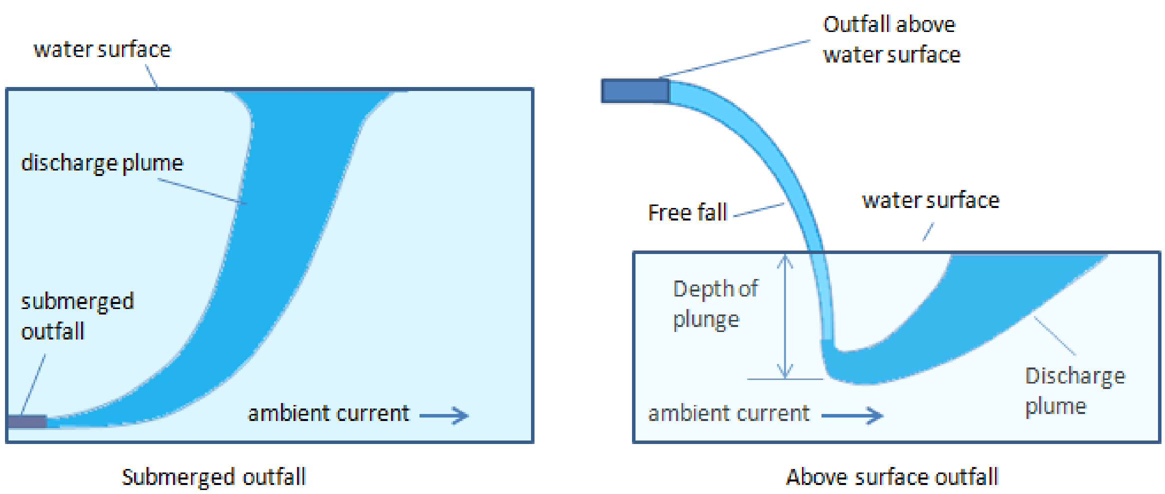

Figure C-6. Submerged and above surface discharge

b. **CORMIX2** deals with submerged multiport diffusers under the same general characteristics as CORMIX1. CORMIX2 analyses unidirectional, staged and alternating multiport diffusers (Figure C-7). The unidirectional diffuser has all ports are more or less normal to the diffuser and pointing horizontally to one side of diffuser. The unidirectional diffuser includes the special case of fanned diffuser (see http://www.cormix.info/CORMIX2.php) and diffuser with ports pointing to the water surface. The latter case is modeled as an alternating port design as it imparts no net horizontal momentum flux to the flow. Staged diffusers have horizontal ports pointing to either side of the diffuser with small deviations, while alternating multiport diffusers have ports pointing to either side of the diffuser in alternating fashion resulting in no net horizontal momentum flux. Alternative diffusers uses the "equivalent slot diffuser" concept and thus neglects the details of the individual jets issuing from each diffuser port and their merging process, but rather assumes that the flow arises from a long slot discharge with equivalent dynamic characteristics. For more details on CORMIX2 see the user's manual (http://www.cormix.info/index.php).

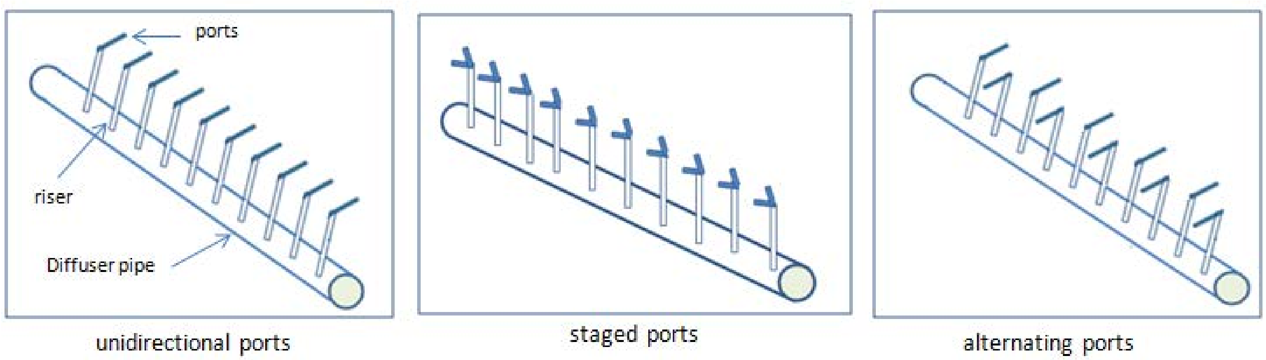

Figure C-7. Some multiport diffuser examples modeled by CORMIX2

c. **CORMIX3** analyzes surface discharges that result when an effluent enters a larger water body laterally, through a canal, channel, or near-surface pipe (Figure C-6). CORMIX3 is limited to positively or neutrally buoyant effluents. However, negatively buoyant surface discharges can be modeled with CORMIX-GTS (http://www.cormix.info/cormix-gts.php). Different surface discharge geometries and orientations can be analyzed using CORMIX3 including flush, protruding and co-flowing channel mouths, and normal, oblique, or parallel orientations to the bank.

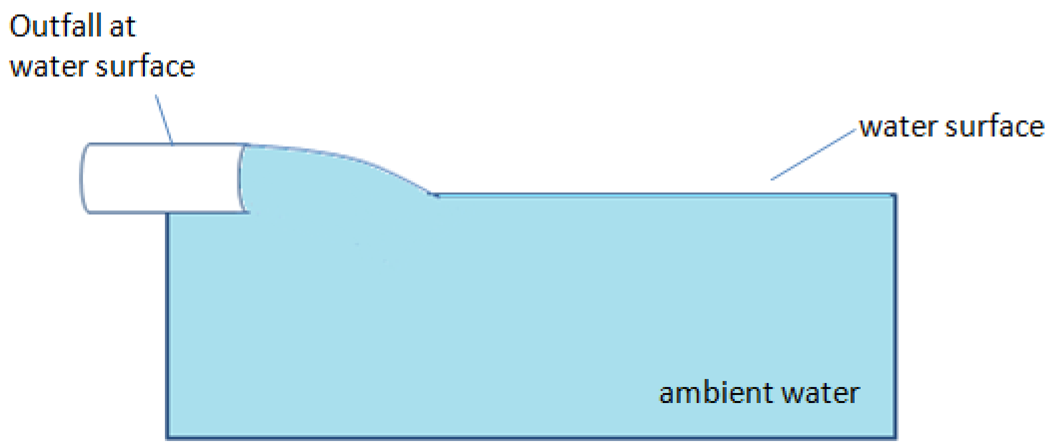

Figure C-8. CORMIX 3 models outfalls discharging through pipes and side channels at the water surface

8.2.5 Visual Plumes

Visual Plumes (VP) is an interface originally developed by EPA that contains the following models: UM3, DKHW, PDSW, and NRFIELD (EPA, 2003). The Visual Plumes 2001 version is still available from EPA's CEAM site. However, updated version is currently available at https://onedrive.live.com/redir?resid=EB2841F2C5C69432%21141. It is likely that future versions will be posted on EPA's CEAM site.

For shallow waters, Visual Plumes may prematurely initiate Brooks (1960) farfield algorithm when plumes reaches water surface (a condition that triggers farfield algorithm in Visual Plumes). Under this condition, the plume may still have both potential and kinetic energy that would allow further discharge induced entrainment before ambient dispersion takes over. This may lead to underestimation of dilution. To overcome this limitation use the procedure outlined in Frick et al. (2010)

a. UM3 is a three-dimensional Updated Merge (UM) model for simulating single and multiport submerged diffusers. Details of the model are available in Frick et al. (2003). UM3 is based on earlier models UOUTPLM (vintage 1979) and UMERGE (vintage 1985) which were two-dimensional models. UM3 uses the 3/2 power profile to calculate the ratio and determine the centerline concentration as a function of the top hat concentration that it predicts. The ratio changes continuously with each integration step along the trajectory (EPA, 1994). Merging is simulated with the reflection technique (Turner, 1970). One shortcoming of UM3 is its inability to recognize and address lateral boundary constraints.

   For diffusers with opposing ports users of UM3 often assume that the diffuser is configured with all ports on one side, the downstream side, creating a co-flowing situation, the port spacing being cut by half. The counter-flowing situation resulting in cross-diffuser merging, can be, but is not simulated explicitly. The simulation offered by UM3 will be quite good if the Roberts' Froude number (F) is > 0.1 because at this current speed the plumes from opposite sides of the diffuser merge rapidly. For a definition and formulation for Roberts' Froude number please see EPA (2003)

Another approach for opposing port diffuser in either fresh or marine water involves simulating only downstream ports. However, it is best used with paired port configurations. This necessitates doubling the flow per port (assuming there is an even number of ports in the diffuser) and increasing the diameter of the ports to maintain approximately the same densimetric Froude number. With this approach only the downstream ports would be used when determining spacing and number of ports. This method may give better simulations if the Roberts' Froude number (F) is < 0.1.

For a diffuser with multiports on each riser, the preferred approach is using an equivalent diffuser with uniformly distributed single ports. The equivalent spacing and effective diffuser lengths are discussed in detail in user manual on Visual Plumes.

b. DKHW is an acronym for Davis, Kannberg, Hirst model for Windows and is a three-dimensional model for submerged single or multi-port diffusers. The basis for the model is described in detail in Davis (1999). It is currently limited to positively buoyant plumes. It considers either single or multiport discharges at an arbitrary horizontal angle into a stratified, flowing current. The current speed and density can vary with depth. It terminates when the surface is reached, the plume reaches its maximum rise height, or when errors are encountered.

   DKHW should generate similar predictions to UM3 in those situations where the discharge port(s) are oriented horizontally and parallel to the current. Where the current flows at an angle to the diffuser axis, a reduced spacing should be specified as outlined in the user's manual (EPA 1994). The effective spacing distance for merging would be the product of the physical spacing and the sine of the angle between the diffuser axis and the current direction. Where the angle is less than 20 deg (approaching parallel current), the angle is set to 20 deg. The latter limit is based on empirical evidence. This is not done dynamically in DKHW, as it is in UM3.

c. PDSW is an acronym for Prych, Davis, Shirazi model for Windows. It is a three-dimensional model that applies to surface discharges of buoyant plumes and includes the effect of surface heat transfer. This is also described in detail in Davis (1999).

d. NRFIELD is based on the RSB model or the Roberts, Snyder, Baumgartner model described in Roberts et al. (1989 a, b, and c). This is an empirical model for multiport diffusers (T-risers, each having two ports for a total of 4-ports) in stratified currents (see Figure C-9). RSB is an updated version of ULINE which is based on experimental studies of multiport diffusers in marine water as described in Roberts (1991). It simulates opposing-port diffusers but predictions are valid if the model runs within the range of the experiments (see EPA 1994). A stand-alone version of RSB (NRFIELD1) is also available (https://onedrive.live.com/redir?resid=CAD9D7A8D73D2995%211724).

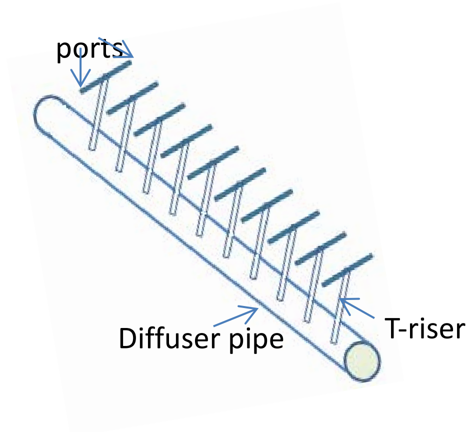

Figure C-9. T-risers with ports discharging on opposite sides of the diffuser pipe

e. VSW stands for Very Shallow Water. It is applicable when the depth approaches three pipe diameters - or less. VSW is a special case of the original two-dimensional UM model. It was originally developed to estimate dilution in very shallow waters for the City of Sumner wastewater treatment plant discharge into White River. It operates out of the original 3PLUMES interface and is available within Visual Plumes. VSW employs the reflection technique (Turner, 1970), which is the same algorithm employed by UM to simulate merging of multiple plumes. Guidance for using 3PLUMES (or the DOS –PLUMES-guide) is available at https://onedrive.live.com/redir?resid=EB2841F2C5C69432%21141. VSW has limited application with a last known application in 1993 for City of Sumner wastewater treatment plant discharge. In 1993, the depth of water near the diffuser was estimated at 0.7 m at under 7Q10 flows. However, minimum dilution based on utilizing 25% of the 7Q10 flows overruled any model predictions and the results of VSW were not used. Other uses of VSW are not known.

9. Dynamic Models

a. **Estuaries:** Application of three-dimensional modeling has been used in estuaries to predict both nearfield and farfield dilution. These models can simulate continuous unsteady flows and treats the ambient waterbody as a very fine 3-D mesh. The scale of the mesh is of the same order as the dimension of the discharge plume. Therefore, these models are time intensive requiring powerful computers and long simulation time. One such model that has been used for mixing zones is Computational Fluid Dynamics (or CFD) model. Fangbiao et al. (2003) cautions that CFD may have limitations in rivers with low flow. Additional information on CFD is available at http://www.cfd-online.com.

b. **Freshwater:** One of the dynamic modeling techniques recommended in EPA (1991) is the use of continuous simulation models. In this approach, daily dilution factors can be estimated from daily ambient and effluent flows. A probability distribution of dilution is then generated which can be used to estimate dilution at the desired averaging period and recurrence interval. One such method that has been used to predict farfield dilution in Washington State is through the use of RiverPlume6 to applicable waterbody (see applicability of RiverPlume6 in the *Farfield* section). Daily ambient and point source discharges are used in RiverPlume6 and corresponding dilution factor are stored as output for 10 or more years. Each of these dilution factors are compared to the maximum dilution allowed (see section on *Minimizing Mixing Zones*). The minimum of these two dilution factors is then stored as a time series. For acute dilution factor the minimum daily dilution is selected for each of the five years and for chronic a minimum 4-day average is selected for each of the years. Cumulative frequency distribution plot is then established and a minimum 1-day and minimum 4-day average dilution factor is then established with a recurrence interval of once every ten years. This dynamic approach using RiverPlume6 is explained further in Section 4. A similar approach may be developed using other steady-state models.

10. Boundary Condition(s)

Boundary conditions are side, surface, and/or bottom constraints which interfere with entrainment of receiving water into the plume. Banks, levees, docks, shallow water, port(s) discharging directly on the bottom, and confined embayments are all examples. The concern is whether the model will reflect these interferences accurately by limiting the entrainment. An additional consideration is whether the constraints are more likely to affect initial dilution or farfield entrainment.

If side boundaries are in close proximity such that initial dilution entrainment is likely to be affected, then for modeling, CORMIX should be used exclusively. However, site-specific computational or analytical options may be available on a case-by-case basis.

Side boundaries may become interferences in the farfield phase of the plume, such as when the plume attaches to the bank downstream in a unidirectional river or stream. Then it may be appropriate to use RiverPlume6, if the attachment (or close proximity) affects horizontally transverse spreading of the mixed effluent.

Plumes that surface inside one or both of the two regulatory boundaries are a common occurrence in estuarine receiving waters because of the additional buoyancy. The surface is the one boundary condition that all models signal decisively.

11. Understanding Farfield Theory

It is reasonable to always assume that the plume’s motion in the ambient receiving water is turbulent. Spreading takes place much faster in turbulent flow than in laminar flow. Farfield begins with gravitational collapse (also referred to as buoyant spreading or density current). This is characterized by lateral spreading of the plume along the layer boundary while it is being advected by the ambient current. Plume thickness probably decreases during this phase; the mixing rate is relatively small.

Following gravitational collapse, the remainder of farfield mixing is best explained by either the theory of turbulent diffusion or shear flow dispersion. Turbulent diffusion employs the turbulent mixing equation of Brooks (1960), wherein the coefficient describing the rate of spread of the plume increases with the size of the plume. The best known facet of this theory is the celebrated "4/3 Power Law" - which says that the diffusion coefficient is proportional to the 4/3 power of the size of the plume. The 4/3 Power Law is described by R.A. Grace (1978). In reality, the Law only applies in homogeneous turbulence far from any boundaries.

Shear flow dispersion employs the longitudinal dispersion equation of Taylor (1954) by the method of Fischer et al. (1979). The theory common to all shear flow is that spreading in the direction of flow is caused primarily by the velocity profile in the cross section. The mechanism Taylor analyzed is often referred to as the "shear effect". It gives a reasonably accurate estimate of the rate of longitudinal dispersion in rivers, and a partial estimate of longitudinal dispersion in estuaries.

11.1 Choosing a Farfield Model

RSB and CORMIX account for gravitational collapse. This phenomenon was observed during the tow-tank experiments, and plume performance during this phase was measured and factored into the empirical equations. Gravitational collapse is not accounted for in the three theoretical, initial dilution models (UM3, UDKHW) or the two farfield models (FARFIELD, RiverPlume6) discussed in this section. Visual Plumes interface allows the modeler to choose a diffusivity coefficient for lateral spreading of the plume in farfield region. Dye tracer studies may be used to quantify site specific diffusivity coefficients for Visual Plumes.

11.1.1 FARFIELD

FARFIELD (see *PermitCalc* here: https://ecology.wa.gov/Regulations-Permits/Guidance-technical-assistance/Water-quality-permits-guidance) is an Ecology spreadsheet that can be used to calculate dilution using the method of N.H. Brooks (1960). This spreadsheet is recommended by Frick et al. (2010) in lieu of using farfield predictions by Visual Plumes (VP) since VP currently does not contain linear diffusivity recommended for estuaries. It should be noted that VP models (except NEARFIELD or RSB) do not account for gravitational collapse described above, although a preliminary solution is provided in Frick et al. (2010). And, Brooks equations only account for lateral diffusion. However, it should be noted that the lateral mixing coefficient is almost 10 times that of vertical diffusion (Fischer et al. 1979). Bricker and Nakayama (2007) studied the farfield horizontal and vertical turbulent diffusion coefficients in an estuary from a concentration field of a wastewater plume and found that the horizontal turbulent diffusion coefficients were two orders of magnitude greater than the vertical diffusion coefficient. Thus, lateral spreading dominates in the farfield region.

Future versions of Visual Plumes may incorporate the FARFIELD spreadsheet. This spreadsheet contains:

- Brook's exponential diffusivity (4/3 power law) for open ocean;
- Brook's linear diffusivity for estuaries; and

- Brook's constant (eddy) diffusivity for rivers.

User instructions for the input are available within the spreadsheet. However, information from model output at the end of the nearfield mixing is required for using this spreadsheet. The user does not need to enter or change any values or formulas in the Output Section. The spreadsheets calculate dilution along the trajectory of the plume and at the specified mixing zone boundary. Optional calculation of pollutant concentrations assuming first-order decay rates is also provided.

11.1.2 RiverPlume6

RiverPlume6 is a two-dimensional model and does not include any nearfield mixing that occurs due to jet velocity or buoyancy of the effluent discharge. Therefore, it is really a farfield model. However, if the plume is completely vertically mixed within a short distance, the model may be applied to both nearfield and farfield predictions. This model is for a neutrally buoyant discharge with no significant discharge-induced mixing. RiverPlume6 should not be used for marine waters due to effluent buoyancy and unsteady (tidal) ambient velocity. The spreadsheet model calculates dilution using the theory of Taylor (1954) by the method described in Fischer et al. (1979) and referred to in EPA's Technical Support Document (1991). The model has been validated using six tracer studies (Cosmopolitan Engineering Group, 2009). It is a one-dimensional model that calculates dilution at a specified point of interest downstream in a river. The calculation for dilution factors incorporates the boundary effect of shorelines using the method of superposition (Fischer et al. 1979, equation 5.9).

This model is based on the assumption that the discharge is: (1) a single point source, which is most appropriate for single port or short diffusers, or side-bank discharges; multiport diffusers in shallow waters may be modeled using principles of superposition; and (2) completely and rapidly mixed vertically, which usually only occurs in shallow rivers. If the diffuser length occupies a substantial portion of the stream width, or the discharge is not vertically mixed over the entire water column within the acute mixing zone, an alternative model should be used. The validity of procedure used by Fischer et al. (1979) is based on the assumption that velocity at any point along a cross-section is equal to the mean cross-sectional velocity. That is, there is no significant transverse variation in current velocity. This assumption is acceptable when a channel is relatively wide. The length scale that is important then is the depth of the stream. The spreadsheet also includes optional calculation of the effective origin of a wastewater source. This is the preferred method. A dye study may be used to substantiate the default lateral dispersion coefficient. Detailed instructions for user input and the spreadsheet model are available within "PermitCalc Workbook" available under Permit Tools at the following internet site: https://ecology.wa.gov/Regulations-Permits/Guidance-technical-assistance/Water-quality-permits-guidance.

11.1.3 CORMIX

In CORMIX the transition between near and farfield predictions is continuous with no distinct break like UM3. Following the completion of the nearfield mixing phase, farfield mixing in CORMIX is simulated with a density current region followed by a passive diffusion region or just a passive region if the plume is non-buoyant. The density current region is characterized by dynamic horizontal spreading and gradual vertical thinning of the mixed effluent flow while being advected by the ambient current. Gravitational collapse is accounted for in CORMIX.

Thus, both lateral and vertical diffusion is allowed. In the passive diffusion region, the dilution is controlled by the turbulent mixing action of the flowing ambient water body.

CORMIX also contains a farfield plume locator (FFLOCATR) algorithm that simplifies schematization of farfield plumes at larger distances within actual flow patterns in natural rivers and estuaries based on the cumulative discharge method. More information is present at the following web site: http://www.cormix.info/picgal/farfield.php.

12. Conducting a Tracer Study

There are four primary objectives that justify conducting a tracer study:

- Confirm the presence of an eddy.
- Quantify dilution.
- Quantify farfield accumulation (reflux).
- Develop a farfield diffusivity or dispersion coefficient.

It is advisable to conduct a reconnaissance survey before the main field work. If the receiving water is tidally-influenced, then the survey should be conducted at the same time in the neap or spring tide cycle and covering the same stages of tide as will be covered during the dye injection. Consideration should be given to deploying a meter to record time, current speed and direction, and depth of water during the survey in order to develop a thorough understanding of anomalies that may be occurring between published tide data and actual field data. Consider taking a cross-section of the channel bottom, if appropriate. These data will allow accurate times to be established for dye injection and measurement. It will also afford an opportunity to set up and run some preliminary cases; which, in turn, will provide some early estimates of plume performance, e.g., trapping depth and horizontal distance to the end of initial dilution.

Concentration of dye in effluent, total loading, and duration of injection deserve careful consideration. Each varies in importance depending upon the objectives of the study, and is discussed below for each objective. A draft plan of study explaining the methods and QA/QC to be employed should be submitted to the Department for review and approval following the reconnaissance survey and prior to initiation of the study.

12.1 Confirm the Presence of an Eddy

If the objective of the study is to simply confirm the presence of an eddy, then concentration, loading, and duration of injection are all relatively unimportant. It is only necessary that the path of the plume can be traced. If, on the other hand, it is necessary to know the mixing ratio in the eddy in order to determine its contribution, then concentration, loading and duration may all be important. If the effluent plume can be seen at the water surface, photographs (particularly aerial photos) are very useful.

12.2 Quantify Dilution

a. **Using a dye**

It may be important to validate a model using a dye study and one set of conditions. This may be the only feasible alternative if critical condition scenarios that need to be examined are quite different from the set of conditions present during the dye study (e.g., future growth). Also, a dye study may be the only reliable way to quantify dilutions if boundary constraints are such that all model results will be suspect. Calibration and validation may serve to increase confidence in model performance.

The most appropriate location to take the measurements for comparing dilutions from the dye study and a model is at the end of the hydrodynamic mixing zone, particularly if the end follows rapid surfacing of the plume in shallow water. However, in some instances both acute and chronic zones should be monitored. The best way of determining the location is via a reconnaissance survey in conjunction with preliminary model runs, using the set of conditions that will be encountered during the field work.

In this situation, a near constant concentration in effluent and duration of injection are important. Total loading is relatively unimportant, *per se*; however, it is desirable to have effluent flow maximized during dye tracer studies, but it is more important to conduct studies at or near critical ambient conditions (e.g. 7Q10 flows or neap tide). In this case, dye tracer study results may need to be adjusted to represent critical effluent flow rates. In tidally-influenced waters, injection and measurements should begin soon after the start of a Lower Low Water slack at sea level during a neap tide or soon after a small flood if it is riverine. This affords the best opportunity to capture a critical condition scenario.

Note: It can be assumed that upstream protrusion does not occur whenever the Roberts' F is > 0.1.

b. **Using other tracer**

In the marine environment, salinity may be used as a tracer to estimate dilution at the edge of the mixing zone. Salinity (conductivity) can also be a useful tracer in freshwater if the effluent is brackish. Cosmopolitan Marine Engineering (2014) compared a UM3 model-predicted dilution factor (65) to that of the salinity-based dilution factor (67) and found good agreement between the two. Salinity was measured both in the background (30 ppt) and at the edge of the mixing zone (average of 29.55) within the plume and the dilution factor was estimated as a ratio of background salinity to the difference in salinity (30/(30 – 29.55)).

12.3 Quantify Farfield Accumulation (Reflux)

This objective warrants considerable discussion because it is difficult to accomplish. Tidal currents may cause effluent to accumulate in the receiving water surrounding an outfall in a tidal river or estuary. The receiving water may also contain background concentrations of pollutants from sources other than effluent. Various methods are available to account for the accumulation of effluent and ambient background sources when determining potential to exceed water quality criteria or estimating wasteload allocations.

There are three methods which are acceptable to Ecology. Two of the methods involve a dye study. Total loading and duration are important factors in both methods; concentration in the effluent can vary during application. The third method involves simply accepting a default value for reflux in lieu of conducting a dye study. Detailed guidance for conducting the methods and mass-balance equations follow.

Farfield accumulation of effluent may be estimated based on either of two methods:

- **Method 1**: The USGS superposition method (Hubbard and Stamper, 1972) may be used by injecting the tracer during one tidal day and measuring continuously at a fixed monitoring station to determine maximum concentrations during succeeding days until the tracer is undetectable; or

- **Method 2**: The Jirka method (EPA, 1992) may be used by injecting the tracer over several tidal cycles (usually five or more) until a quasi-maximum steady state is reached. Concentrations of the tracer are usually monitored continuously at a fixed monitoring station.

In addition to two methods of tracer injection, two alternative schemes for locating monitoring stations are acceptable:

Alternative 1: Tracer concentrations are measured in the nearfield at the mixing zone boundary in the approximate centerline of the effluent plume; or

Alternative 2: Tracer concentrations are measured in the farfield at some considerable distance from the effluent plume at a position that is representative of the source of dilution water for the plume.

Either the superposition (Method 1) or Jirka method (Method 2) may be used to conduct the tracer studies for both Alternatives 1 and 2.

- **Method 3**: A third method is also proposed if a tracer study is not conducted. In this method a default correction which can be used as an approximation of farfield accumulation based on recommendations by EPA (1992).

A number of terms which will be used during this discussion need to be defined:

- nearfield: at the chronic mixing zone boundary in the approximate center-line of the effluent plume.
- farfield: at some considerable distance from the effluent plume at a position that is representative of the source of dilution water for the plume.
- V: initial maximum effluent concentration (volume fraction of effluent; e.g., 5 percent effluent corresponds to V of 0.05) during first tidal cycle prior to influence of farfield accumulation from previous tidal cycles.

- $\overline{V}$: quasi-steady-state maximum effluent concentration (volume fraction of effluent; e.g., 5 percent effluent corresponds to $\overline{V}$ of 0.05) after several tidal cycles result in equilibrium with farfield accumulation.
- r_d: return rate of dye or effluent mass discharged in the previous tidal cycle as defined in EPA (1992).
- DF: initial effluent dilution factor (reciprocal of volume fraction of effluent; e.g., 5 percent effluent corresponds to DF of 20) during first tidal cycle prior to influence of farfield accumulation from previous tidal cycles. DF may be estimated using a model (e.g., PLUMES) or by nearfield tracer measurement. DF is usually determined at critical conditions.
- $\overline{DF}$: quasi-steady-state effluent dilution factor (reciprocal of volume fraction of effluent*;* e.g., 5 percent effluent corresponds to $\overline{DF}$ of 20) after several tidal cycles (usually 5 or more cycles) result in equilibrium with farfield accumulation. $\overline{DF}$ is usually determined at critical conditions.
- C_p: pollutant concentration measured as a flux-average value in the plume at the mixing zone boundary. (Refer to section on Average versus Centerline).
- C_e: pollutant concentration in effluent discharged from the outfall pipe.
- C_a: pollutant concentration in upstream ambient receiving water (i.e., away from the influence of farfield accumulation).
- WLA: effluent concentration to use for Wasteload Allocation (acute or chronic) for derivation of water quality-based permit limits.
- WQC: pollutant concentration for water quality criteria (acute or chronic).
- GPS: geographic positioning system.
- CTD profiler: an instrument that provides conductivity temperature and depth profiles

12.3.1 Methods 1 and 2: Alternative 1 using a dye

If the tracer monitoring station is located in the nearfield, then the following mass-balance equations are appropriate:

calculate Jirka's r_d from nearfield $\overline{V}$ and V (based on equation 22 in (EPA, 1992)):

$$r_d = \frac{\left(\overline{V} - V\right)}{\overline{V}} \tag{5}$$

calculate the nearfield $\overline{DF}$ (acute or chronic boundary), including the effect of farfield accumulation of effluent, from model or tracer estimates of DF and estimated r_d in the previous step (based on equation 22 in (EPA, 1992)):

$$\overline{DF} = DF\left(1 - r_d\right) \tag{6}$$

The following equation is appropriate to calculate pollutant concentrations (C_p) at the mixing zone boundaries for comparisons with water quality criteria. Nearfield dilution is corrected for farfield accumulation of effluent in the previous step. The following equation incorporates the effect of ambient background (C_a) from sources of pollutants other than effluent. Estimates of C_e may also include a reasonable potential multiplier using methods in Chapter 6 of this Manual.

Pollutant concentrations (C_p) are estimated as follows (based on equation 9 in (EPA, 1994):

$$C_p = C_e\left(\frac{1}{\overline{\overline{DF}}}\right) + C_a\left(1-\left(\frac{1}{\overline{\overline{DF}}}\right)\right) \tag{4a}$$

calculate acute and chronic WLAs:

$$WLA = WQC * \overline{DF} - C_a\left(\overline{DF} - 1\right) \tag{7}$$

Example**:**

Given: nearfield V = .02 (2 percent effluent); nearfield $\overline{\mathrm{V}}$ = .07 (7 percent effluent).

Calculation of nearfield $\overline{\mathrm{DF}}$ including farfield accumulation of effluent:

$$r_d = \frac{(.07-.02)}{.07} = .7143;\; DF = \frac{1}{.02} = 50;\; \text{therefore, nearfield } \overline{DF} = 50(1-.7143) = 14.3$$

12.3.2 Methods 1 and 2: Alternative 2 using a dye

If the tracer monitoring station is located in the farfield, then the following mass-balance equations are applicable:

calculate nearfield DF, excluding the farfield accumulation of effluent, from a mixing zone model or from an additional nearfield tracer monitoring station (e.g., nearfield DF = reciprocal of nearfield V)

calculate the nearfield $\overline{\mathrm{DF}}$ (acute or chronic boundary), including the effect of farfield accumulation of effluent, by mass balance with nearfield DF from the previous step and farfield $\overline{\mathrm{V}}$ (based on equation 8 in (EPA, 1994)):

$$\overline{DF} = \frac{DF}{\left(1+\overline{V}(DF-1)\right)} \tag{8}$$

The following equation is appropriate to calculate pollutant concentrations (C_p) at the mixing zone boundaries for comparisons with water quality criteria. Nearfield dilution is corrected for farfield accumulation of effluent in the previous step. The following equation incorporates the effect of ambient background (C_a) from sources of pollutants other than effluent. Estimates of C_e may also include a reasonable potential multiplier using methods in Chapter 6 of this Manual. Pollutant concentrations (C_p) are estimated as follows (based on equation 9 in (EPA, 1994)):

$$C_p = C_e\left(\frac{1}{\overline{\overline{DF}}}\right) + C_a\left(1-\left(\frac{1}{\overline{\overline{DF}}}\right)\right) \tag{4a}$$

calculate acute and chronic WLAs:

$$WLA = WQC * \overline{DF} - C_a\left(\overline{DF} - 1\right) \tag{7}$$

Example:

Given**:** nearfield DF=50 from PLUMES model excluding farfield accumulation of effluent; farfield $\overline{\mathrm{V}}$ =.051 (5.1 percent effluent) from tracer study using super-position method.

Calculation of nearfield $\overline{\mathrm{DF}}$ including farfield accumulation of effluent**:**

$$\text{nearfield } \overline{DF} = \frac{50}{\left(1 + .051(50 - 1)\right)} = 14.3$$

12.3.3 Method 3

1) **Using a default farfield accumulation factor**

If it is decided to use a default correction for farfield accumulation, then the following mass balance equations are applicable:

estimate default for Jirka's r_d = 0.5 from EPA (1992).

calculate the nearfield $\overline{\mathrm{DF}}$ (acute or chronic boundary), including the effect of farfield accumulation of effluent, from model or tracer estimates of DF and estimated r_d in the previous step (based on equation 22 in (EPA, 1992)):

$$\overline{DF} = DF\left(1 - r_d\right) \tag{6}$$

The following equation is appropriate to calculate pollutant concentrations (C_p) at the mixing zone boundaries for comparisons with water quality criteria. Nearfield dilution is corrected for farfield accumulation of effluent in the previous step. The following equation incorporates the effect of ambient background (C_a) from sources of pollutants other than effluent. Estimates of C_e may also include a reasonable potential multiplier using methods in Chapter 6 of this Manual. Pollutant concentrations (C_p) are estimated as follows (based on equation 9 in (EPA, 1994):

$$C_p = C_e\left(\frac{1}{\overline{\overline{DF}}}\right) + C_a\left(1 - \left(\frac{1}{\overline{\overline{DF}}}\right)\right) \tag{4a}$$

calculate acute and chronic WLAs:

$$WLA = WQC * \overline{DF} - C_a\left(\overline{DF} - 1\right) \tag{7}$$

Example:

Given: r_d=0.5; DF=50

Calculation of $\overline{DF}$: $\overline{DF} = 50(1 - 0.5) = 25.$

2) Using drogues and salinity profiles in marine waters

Drogues may be deployed near the diffuser at several depths and their trajectories mapped with GPS over tidal cycles to establish whether water near the diffuser would return back to the general vicinity of the diffuser. In addition CTD profiles over tidal cycle near the diffuser would show if down-welling occurred at the same time to bring back the refluxed effluent back to the diffuser. This procedure is presented in detail in Cosmopolitan Marine Engineering (2014). As an example, if background salinity was 30.33 ppt and the CTD profile showed a salinity of 30.12 ppt at the depth of the diffuser away from the discharge plume. This indicated that part of the plume that surfaced down welled to the diffuser depth. The reflux amount was 0.7% ((30.33 - 30.12)/30.33). This concentration was then added to the background and model rerun to estimate a reflux based dilution factor.

3) Using theoretical models with tidal buildup capability.

a) CORMIX considers reduction in initial dilution due to re-entrainment of material remaining from the previous tidal cycle. It does not consider unsteady buildup of material over several tidal cycles. For procedure on how to use this capability refer to the CORMIX user's manual available at: http://www.mixzon.com/downloads/.

b) Visual Plumes includes a tidal buildup option in which time-series data for current velocity and direction must be input. An output plot is created that shows decrease in effective dilution as background concentrations increase with tidal reversals and associated re-entrainment of discharge plume material. To use this capability refer to Visual Plumes user's manual (EPA 2003). This capability in Visual Plumes should not be used as a substitute for actual measured pollutant data (for example a dye). This capability is useful when measurements are unavailable. The tidal buildup capability does not include dispersive or other effects that will influence pollutant concentrations in tidal waters.

c) Complex hydrodynamic models such as the three dimensional Generalized Environmental Modeling System for Surfacewaters (GEMSS), used to develop South Puget Sound model (Roberts et al. 2014a; and Ahmed et al. 2014), and Finite Volume Coastal Ocean Model (FVCOM), used to develop the Salish Sea model (Roberts et al. 2014b); or Water Quality Analysis Simulation Program (WASP), used to develop the Willapa Estuary model; may be used to establish reflux.

12.4 Develop a Farfield Diffusion Coefficient

a. Establish a uniform tracer concentration in the effluent.

b. Conduct multiple transects across the entire width of the plume.

c. Conduct multiple profiles from above plume to below plume, plot data in 2-D or 3-D to assess whether a Gaussian shape is observed.

d. Run model (VP or RiverPlume6) at variable dispersion values to obtain best fit to observed data. This is most feasible in 2-D if the profiles (from c.) show that the plume is fully mixed vertically. But it is still possible to do this for 3-D plume.

e. Steps b and c can be combined by raising or lowering the fluorometer in a sawtooth pattern.

13. What to look for in a Mixing Zone Study

A mixing zone study should include the following at a minimum.

13.1 General Requirements

1. A statement confirming that AKART has been applied by the entity seeking a mixing zone.
2. A description of the maximum size of the mixing zone allowed under the regulations.
3. An analysis showing how mixing zones have been minimized based on using the lowest dilution from hydraulic limitation, width limitations, distance limitation and that predicted by the model.
4. An evaluation showing no environmental harm from the proposed diffuser location and requested mixing zone. This should include a conclusion that there will be no damage to the ecosystem, nor loss of sensitive habitat; and that there will be no adverse public health effects, and no interference with existing or characteristic uses of the waterbody. Mixing zone analyses for new outfalls or reconstructed outfalls that are located on state-owned aquatic lands, or where the outfall pipe runs through state-owned aquatic lands, should describe how the proponent has contacted the Department of Natural Resources (DNR) to determine potential habitat issues and mitigation requirements. Include a copy of any reports or correspondence with DNR documenting the results of the determination.
5. A clear description of the critical conditions used for dilution factors:
   a. For ambient freshwater (unidirectional flow) use 7Q10 flows for acute, chronic and non-carcinogen pollutants, and harmonic flow for carcinogens.
   b. For ambient marine waters (and reversing flows e.g., tidally-influenced rivers) use 10^{th} or 90^{th} percentile current velocity for acute and 50^{th} percentile tidal current velocity for chronic, carcinogens and non-carcinogens.

c. Generally, use depth of outfall at 7Q10 flows (rivers) or at MLLW (marine environment). For assessing human health in freshwater, depths of outfall should be established at the applicable flow (e.g. harmonic mean flow or 30Q5 flows). For tidally influenced rivers a combination of MLLW and critical river flows should be used to establish depth of outfall.
d. Use density profile that gives the lowest dilution. Valuate both maximum and minimum stratification. For human health, use average density profiles to estimate dilution.
e. For unidirectional flow use centerline dilution factor for acute and chronic conditions, while flux average for human health dilution factors. For marine environment or rivers with reversing flows, use flux-average dilution factors for all conditions.

6. For over lapping mixing zones ensure that the maximum mixing zone size limitations are not exceeded, while also ensuring no environmental harm. (see 4 above)
7. For extended mixing zones, if proposed, must ensure that altering the size increases protection, volume of effluent is more beneficial than removing it, that the effluent is necessary for social or economic development for the area, and the discharge existed prior to 1992.

13.2 Diffuser Information

1. Location, orientation, description and dimension of diffusers and ports
   a. Latitude, longitude and/or river mile
   b. Single port or multiport (opposing ports or ports on same side)
   c. Number or ports, port spacing and port diameter
   d. Diffuser lengths and angles. Angles in CORMIX are based upon current direction and horizontal plane. In Visual Plumes angles are based on geographical datum and horizontal plane (Figure C-10).

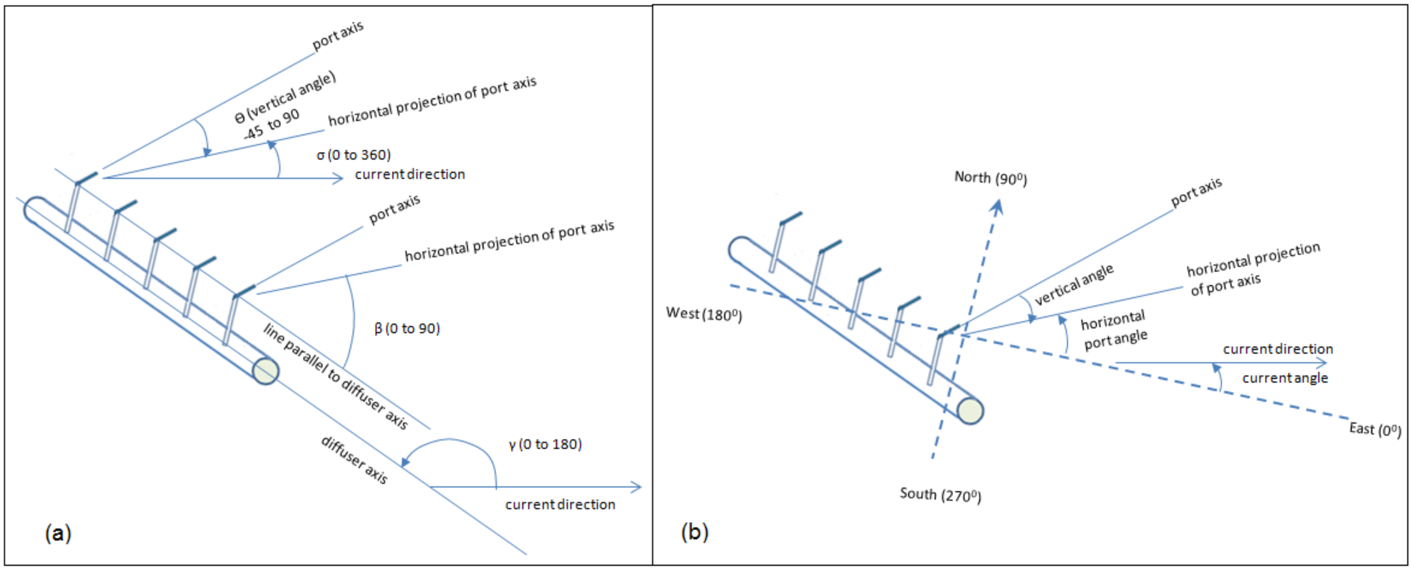

Figure C-10. Diffuser and port orientation angles (a) CORMIX and (b) Visual Plumes

2. Port elevation above bottom and the depth of the diffuser/port below water surface based on either 7Q10 flow (for rivers) or MLLW (for marine or tidally-influenced river reaches).
3. Plan view maps showing the mixing zone size and dimensions in relation to the diffuser.
4. Schematic of waterbody cross-section, showing diffuser location in relation to shoreline and bottom.
5. Report on the integrity of the diffuser and the ports being modeled.

13.3 Discharge Characteristics

- Maximum daily flows (acute) or maximum monthly averages (chronic), or design storm flows.
- Discharge density (temperature and salinity).
- Pollutant characteristics, human carcinogens, non-carcinogens, aquatic life toxicity, etc.

13.4 Ambient Water Characteristics

- Critical stream flow statistics (7Q10, 30Q5, harmonic flow) or marine current velocities (10^{th}, 90^{th} and 50^{th} percentiles over a neap and spring tide and directions).
- Velocity profile in the vicinity of the diffuser.
- Temporal density (temperature and salinity) profiles near the diffuser. May need to consider both seasonal and tidal variability.
- Manning's roughness coefficient, if used.
- Schematic of cross-section showing channel width, depth and location of diffuser.

13.5 Model

- Model selection and application discussion. Consider model applicability to single or multiport diffuser, opposing port configuration, submerged, surface or above-surface discharge, buoyant or non-buoyant discharge, and potential plume attachment to boundaries.
- Description of mixing and plume dynamics (nearfield, farfield, tidal buildup/reflux).
- Sensitivity analysis.
- Calibration to empirical data (tracer studies), if necessary.
- Provide model output and summary table of results.

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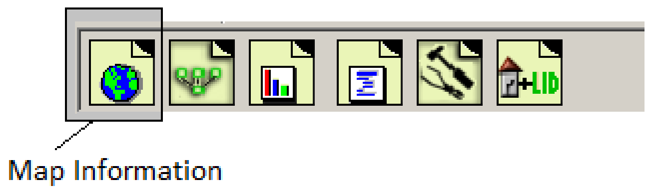
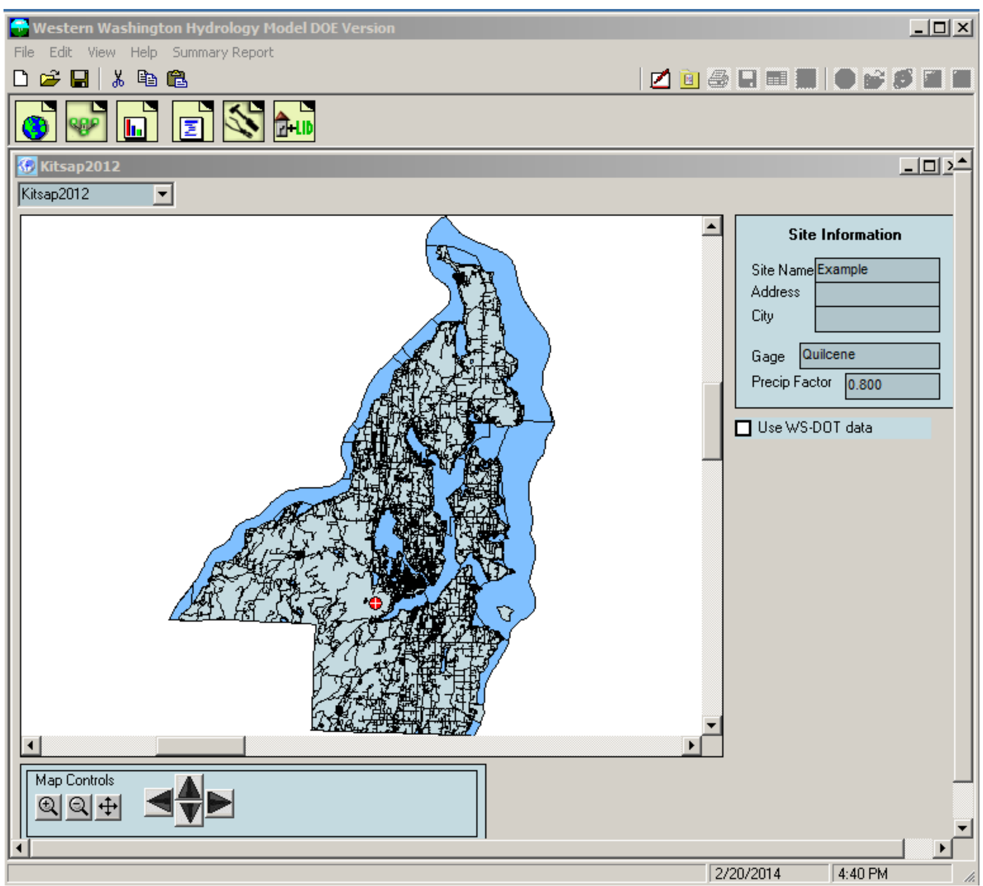
Attachment C.1 Estimating critical stormwater flow for Western Washington

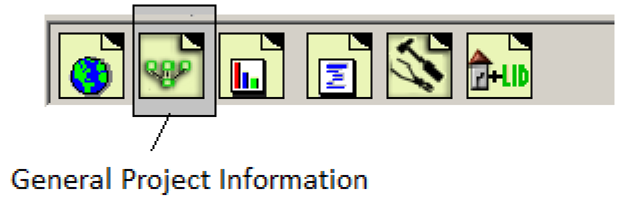
1. Download and install the current version of Western Washington Hydrology Model (WWHM) from Ecology's web site: https://ecology.wa.gov/Regulations-Permits/Guidance-technical-assistance/Stormwater-permittee-guidance-resources/Stormwater-manuals/Western-Washington-Hydrology-Model.

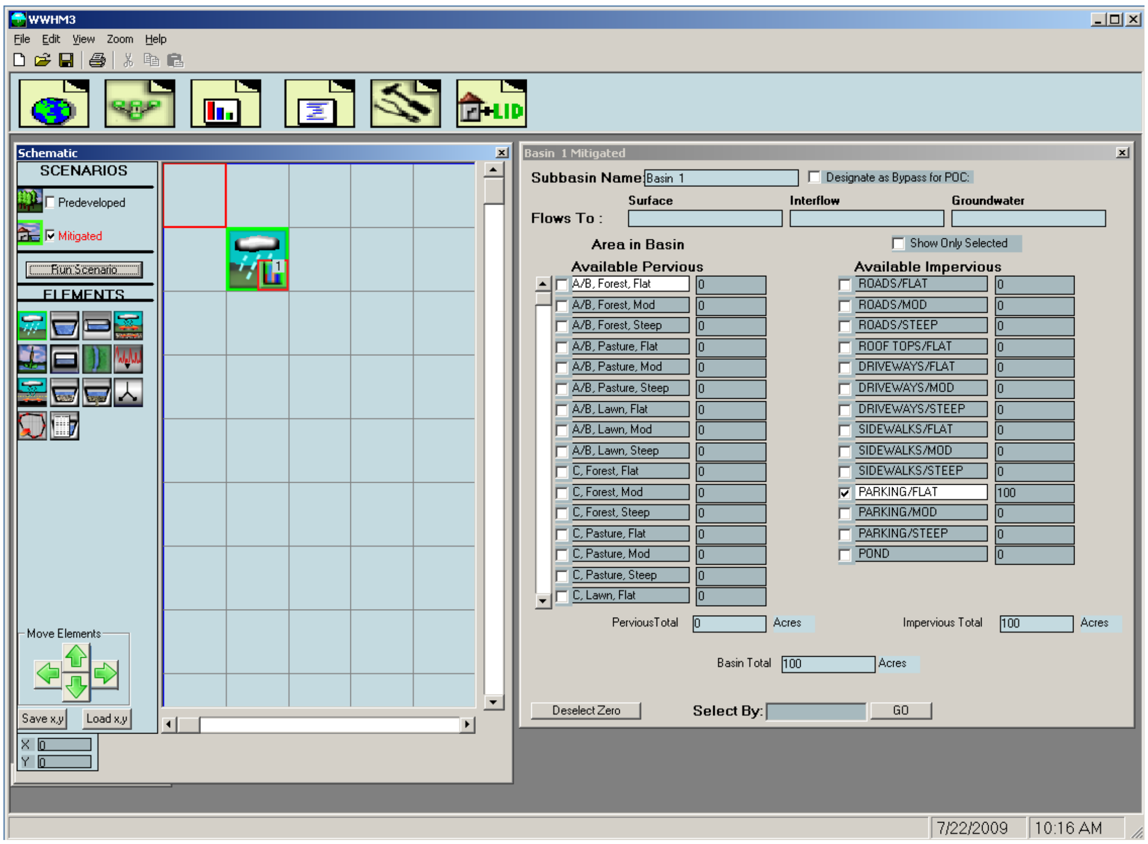
2. Open WWHM and click on the first application toolbar "Map Information" shown below. This is the default window when you open WWHM. Enter the county and site information in the first window. Click on the approximate site location on the county map. This populates the gage and precipitation factor information in the appropriate boxes and the site location is represented by a red circle on the map.





3. Click the next icon (general project information) and the Schematic window opens.





a. Check the “mitigated” box.
b. Drag the land use basin icon (the first one under Basic Elements) into the schematic window.
c. Enter the land use acreage in the appropriate boxes on the right.

4. Right click on the land use icon on the grid and choose “connect to point of compliance” outlet as surface.

5. Click “Run Scenario.”

6. Click on the tool icon (5th from left) 

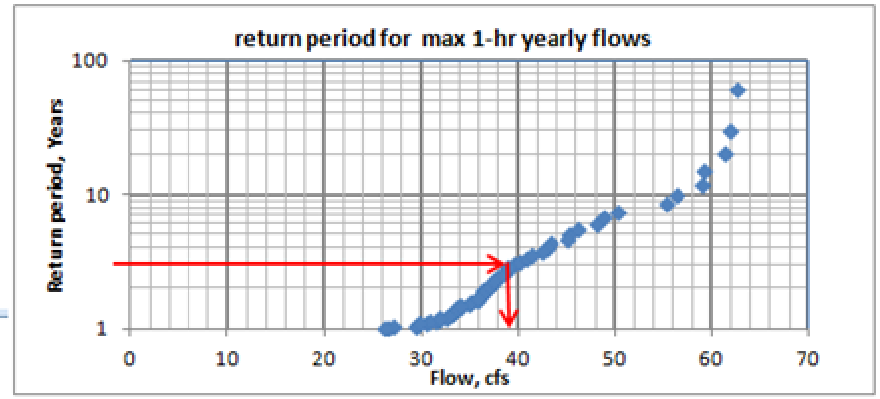
a. Click on Export data and highlight POC1 mitigated flow. Choose hourly average (for acute critical flow) or daily average (for chronic critical flow) as the output time step.
b. Click Export and choose the output format as ASCII or comma delimited.

7. In Excel, import the data and convert text to columns parsing the data to generate date/time and 1-hr (or 24-hr) flow columns. Establish maximum 1-hour (or for chronic, a maximum 4-day average) flow for each of the years in the data set. Remember the data is based on a water year (Oct 1 of previous year to Sept 30 of current year) and if you want to use all data choose water year, if not, choose calendar year, and you may have to delete some data in the beginning and end of the dataset.

8. Sort the maximum yearly hour-flow or 4-day average flow data in descending order and calculate the probability of exceedance for each of the storm flows by ranking each of the data points and dividing by one more than the total number of data points. The return period is simply the reciprocal of the probability of exceedance of the storm flows.

9. Plot the resulting return periods on y-axis with the corresponding flows on x-axis (see below for an example). Flow corresponding to a return period of 3 years is the design storm flow.

	A	B	C	D	E	F	G	H
1	Date	Time	Flow, cfs	year		Year	Max_hourly flow, cfs	return period for 1-hr flow, yrs
2	10/1/1948	00:00	5.31E-03	1948		1949	45.26	5.083
3	10/1/1948	01:00	4.86E-03	1948		1950	36.13	1.794
4	10/1/1948	02:00	4.46E-03	1948		1951	43.29	4.357
5	10/1/1948	03:00	4.11E-03	1948		1952	38.60	2.773
6	10/1/1948	04:00	3.79E-03	1948		1953	42.81	4.067
7	10/1/1948	05:00	3.51E-03	1948		1954	36.28	1.968
8	10/1/1948	06:00	3.26E-03	1948		1955	50.18	7.625
9	10/1/1948	07:00	3.03E-03	1948		1956	29.55	1.089
10	10/1/1948	08:00	2.82E-03	1948		1957	37.13	2.179
11	10/1/1948	09:00	2.63E-03	1948		1958	31.54	1.173
12	10/1/1948	10:00	2.46E-03	1948		1959	39.47	3.050
13	10/1/1948	11:00	2.31E-03	1948		1960	29.30	1.070
14	10/1/1948	12:00	2.16E-03	1948		1961	40.00	3.211
15	10/1/1948	13:00	2.03E-03	1948		1962	37.44	2.346
16	10/1/1948	14:00	1.91E-03	1948		1963	33.67	1.452
17	10/1/1948	15:00	1.80E-03	1948		1964	26.92	1.052
18	10/1/1948	16:00	1.70E-03	1948		1965	48.09	6.100
19	10/1/1948	17:00	1.61E-03	1948		1966	36.69	2.033
20	10/1/1948	18:00	1.52E-03	1948		1967	38.58	2.652
21	10/1/1948	19:00	1.44E-03	1948		1968	33.93	1.488
534709	9/30/2009	11:00	0	2009		2003	61.28	20.333
534710	9/30/2009	12:00	0	2009		2004	46.16	5.545
534711	9/30/2009	13:00	0	2009		2005	62.57	61.000
534712	9/30/2009	14:00	0	2009		2006	58.84	12.200
534713	9/30/2009	15:00	0	2009		2007	33.97	1.525
534714	9/30/2009	16:00	0	2009		2008	37.41	2.259
534715	9/30/2009	17:00	0	2009				
534716	9/30/2009	18:00	0	2009				
534717	9/30/2009	19:00	0	2009				
534718	9/30/2009	20:00	0	2009				
534719	9/30/2009	21:00	0	2009				
534720	9/30/2009	22:00	0	2009				
534721	9/30/2009	23:00	0	2009				



=1/(RANK(G2,$G$2:$G$61)/(COUNT($G$2:$G$61)+1))

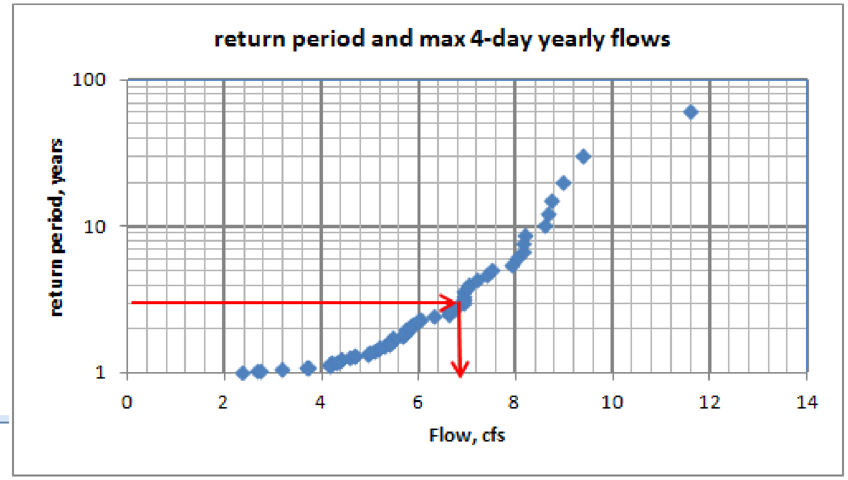
=YEAR(A2)

{=MAX(IF($D$2:$D$534721=F2,$C$2:$C$534721)) }

Use Ctrl + Shift + Enter to execute the nested formula

Figure C-11. Calculations for maximum hourly storm flows with a return period of 3 years

	A	B	C	D	E	F	G	H	I
1	Date	Time	daily flow, cfs	4-day avg flow, cfs	year		Year	Max 4-day flow, cfs	return period for 4-day flow, yrs
2	10/1/1948	00:00	0.0026		1948		1949	5.46	1.74
3	10/2/1948	00:00	0.0007		1948		1950	6.66	2.77
4	10/3/1948	00:00	4.1855		1948		1951	4.19	1.17
5	10/4/1948	00:00	0.6672	1.21E+00	1948		1952	7.02	4.07
6	10/5/1948	00:00	0.0154	1.22E+00	1948		1953	5.07	1.42
7	10/6/1948	00:00	0.0017	1.22E+00	1948		1954	6.61	2.54
8	10/7/1948	00:00	0.9792	4.16E-01	1948		1955	5.72	1.91
9	10/8/1948	00:00	0.0057	2.51E-01	1948		1956	6.93	3.59
10	10/9/1948	00:00	0.0011	2.47E-01	1948		1957	8.18	8.71
11	10/10/1948	00:00	1.5388	6.31E-01	1948		1958	4.39	1.24
12	10/11/1948	00:00	0.0309	3.94E-01	1948		1959	11.58	61.00
13	10/12/1948	00:00	0.0021	3.93E-01	1948		1960	5.27	1.53
14	10/13/1948	00:00	0.0006	3.93E-01	1948		1961	5.74	1.97
15	10/14/1948	00:00	0.0009	8.60E-03	1948		1962	7.20	4.36
16	10/15/1948	00:00	0.0000	8.86E-04	1948		1963	5.40	1.61
17	10/16/1948	00:00	0.0000	3.70E-04	1948		1964	2.66	1.03
18	10/17/1948	00:00	0.0000	2.14E-04	1948		1965	4.18	1.15
19	10/18/1948	00:00	0.0000	0.00E+00	1948		1966	8.60	10.17
20	10/19/1948	00:00	0.0000	0.00E+00	1948		1967	6.29	2.44
21	10/20/1948	00:00	0.0000	0.00E+00	1948		1968	6.91	3.39
22269	9/18/2009	00:00	0.0000	0.00E+00	2009		2003	7.49	5.08
22270	9/19/2009	00:00	0.0000	0.00E+00	2009		2004	3.17	1.07
22271	9/20/2009	00:00	0.2536	6.34E-02	2009		2005	5.70	1.85
22272	9/21/2009	00:00	0.0114	6.62E-02	2009		2006	5.19	1.49
22273	9/22/2009	00:00	0.0015	6.66E-02	2009		2007	5.96	2.26
22274	9/23/2009	00:00	0.0013	6.69E-02	2009		2008	5.78	2.03
22275	9/24/2009	00:00	0.0000	3.53E-03	2009				
22276	9/25/2009	00:00	0.0000	6.89E-04	2009				
22277	9/26/2009	00:00	0.0000	3.20E-04	2009				
22278	9/27/2009	00:00	0.0000	0.00E+00	2009				
22279	9/28/2009	00:00	0.0000	0.00E+00	2009				
22280	9/29/2009	00:00	0.0000	0.00E+00	2009				
22281	9/30/2009	00:00	0.0000	0.00E+00	2009				



=1/(RANK(H2,$H$2:$H$61)/(COUNT($H$2:$H$61)+1))

=YEAR(A2)

{=MAX(IF($E$2:$E$22281=G2,$D$2:$D$22281))}

Use Ctrl + Shift + Enter to execute the nested formula

Figure C-12. Calculations for maximum 4-day storm flows with a return period of 3 years

Attachment C.2 - Estimating steady-state 10th, 50th, and 90th percentile current velocities for estuaries and tidally-influenced river systems

The velocity distribution analysis should be produced from a data set consisting of periodic readings taken by an instrument deployed over a neap and spring tide cycle near or at diffuser depth. Drogues (apparatus that drifts with currents at specified depths with positions recorded at intervals) maybe used as an alternative to current meters. Site specific data should be employed when possible. The following example uses data from a NOAA's station.

Example: Current data (tidal velocity magnitude and direction) is available at Cherry Point at NOAA's PORTS station (ID: cp0101). Data for the month of July is downloaded from NOAA's site.
(https://tidesandcurrents.noaa.gov/cdata/DataPlot?id=cp0101&bin=0&bdate=20180905&edate=20180906&unit=1&timeZone=UTC).

1. Download data into Excel spreadsheet (as shown in Figure C-13). Use only absolute value of the velocity since we are using steady-state models.

2. For each velocity data calculate the cumulative probability of occurrence as shown. This is the "distribution free" (non parametric) method (Martin and McCutcheon, 1999).

3. Plot current velocity versus cumulative probability of occurrence.

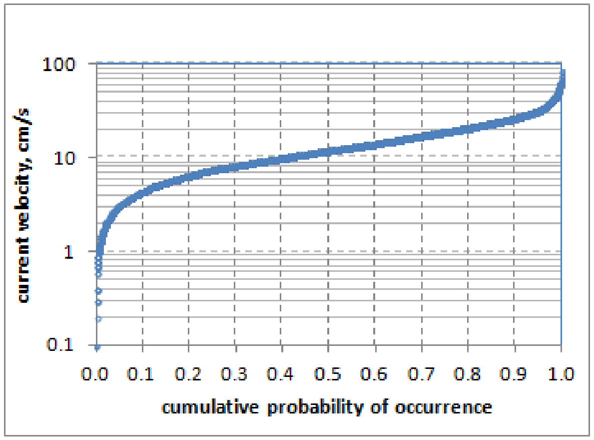
4. Read the 10th, 50th and 90th percentile velocities directly from the plot. For example, 10th percentile velocity will have a 10% probability of exceedance, and so forth.

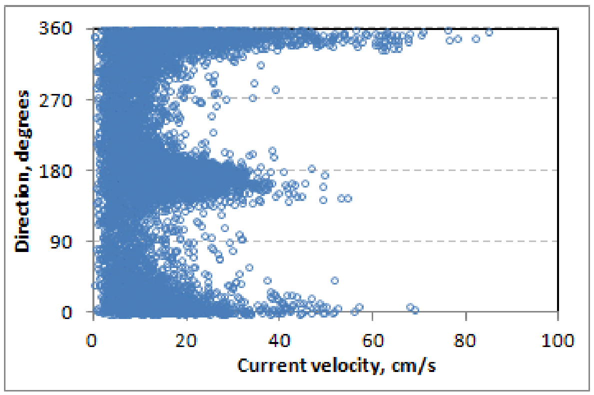
5. Plot current direction versus current velocity to establish predominant direction of the currents. For example, for this dataset, the predominant direction is 0, 180, or 360 degrees, i.e., east-west direction. This is also clear from a "velocity-rose" plot (Figure C-13). The "velocity-rose" plot is similar to the "wind-rose" plot and can be created from any of the available "wind-rose" plotting software.

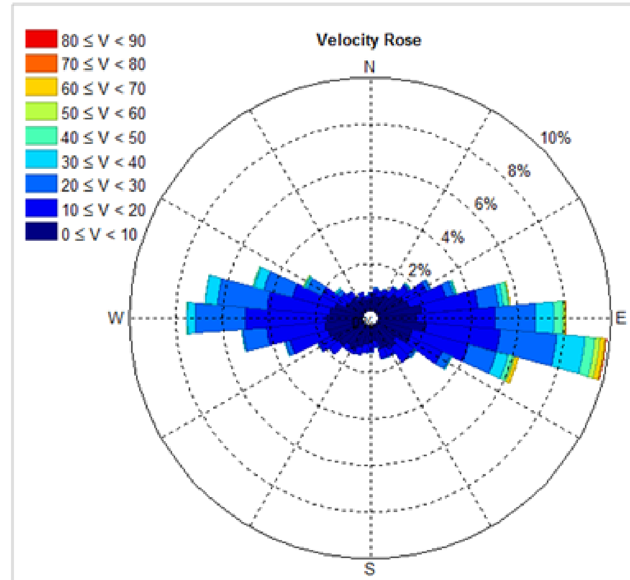
6. For a diffuser port that is oriented towards north (90 degree) the current direction would be either 0 degree or 180 degree based on geographic coordinate system (as in Visual Plumes). In CORMIX the horizontal angle (SIGMA) would be 90 degrees.

	A	B	C	D	E
1	Cherry Point, WA	Date/Time	velocity, cm/s	direction, degrees	cumulative probability of occurrence
2		7/1/2014 0:05	18.1	357	0.714
3		7/1/2014 0:11	18.6	358	0.730
4		7/1/2014 0:17	17.1	1	0.685
5		7/1/2014 0:23	20.8	351	0.787
6		7/1/2014 0:29	23.1	349	0.834
7		7/1/2014 0:35	23.3	346	0.837
8		7/1/2014 0:41	23.2	355	0.836
9		7/1/2014 0:47	24.1	360	0.852
10		7/1/2014 0:53	23.9	353	0.848
11		7/1/2014 0:59	23.8	353	0.847
12		7/1/2014 1:05	24.6	351	0.860
13		7/1/2014 1:11	22.7	356	0.826
14		7/1/2014 1:17	24.8	351	0.864
15		7/1/2014 1:23	26.3	354	0.888
16		7/1/2014 1:29	25	349	0.867
17		7/1/2014 1:35	22.4	352	0.820
18		7/1/2014 1:41	15.7	355	0.644
19		7/1/2014 1:47	16.1	7	0.656
20		7/1/2014 1:53	16	6	0.651
21		7/1/2014 1:59	11.5	19	0.464
22		7/1/2014 2:05	11.3	359	0.455
23		7/1/2014 2:11	11.2	40	0.450
24		7/1/2014 2:17	10.3	37	0.406
25		7/1/2014 2:23	4	44	0.081
26		7/1/2014 2:29	7.4	44	0.239
27		7/1/2014 2:35	10.2	87	0.401
28		7/1/2014 2:41	6.6	48	0.199
29		7/1/2014 2:47	10.2	69	0.401
30		7/1/2014 2:53	16.3	3	0.662
31		7/1/2014 2:59	8.2	52	0.284
32		7/1/2014 3:05	9.7	21	0.372
33		7/1/2014 3:11	8.6	29	0.308
34		7/1/2014 3:17	14.5	82	0.600
35		7/1/2014 3:23	5.2	53	0.128
36		7/1/2014 3:29	7.4	353	0.239
37		7/1/2014 3:35	8.7	357	0.315
7415		7/31/2014 23:41	14.8	345	0.611
7416		7/31/2014 23:47	16.4	343	0.665
7417		7/31/2014 23:53	17.4	340	0.694
7418		7/31/2014 23:59	14.2	344	0.589

Distribution free (non-paraametric) method . page 138. Hydrodynamics and transport for water quality modeling. James L. Martin, Steve C. McCutcheon. 1999. CRC Press, Boca Raton, Florida







cumulative probability =RANK(C7418,$C$2:$C$7418,1)/(COUNT($C$2:$C$7418)+1)

Figure C-13. Analysis of tidal current velocity and direction for establishing 10th, 50th and 90th percentile current velocities for acute and chronic dilution analysis

Attachment C.3 Delineation of Estuaries

WAC173-201A-400(7)(b)(ii) defines areas to the east of a line from Green Point (Fidalgo Island) to Lawrence Point (Orcas Island) are considered estuarine as are all of the Strait of Georgia and the San Juan Islands north of Orcas Island (Figure C-14 (a)),. To the east of Deception Pass, and to the south and east of Admiralty Head, and south of Point Wilson on the Quimper Peninsula, is Puget Sound proper (Figure C-14 (b)), which is considered to be entirely estuarine. All waters existing within bays from Point Wilson westward to Cape Flattery and south to the North Jetty of the Columbia River (Figure C-14 (c)) shall also be categorized as estuarine.

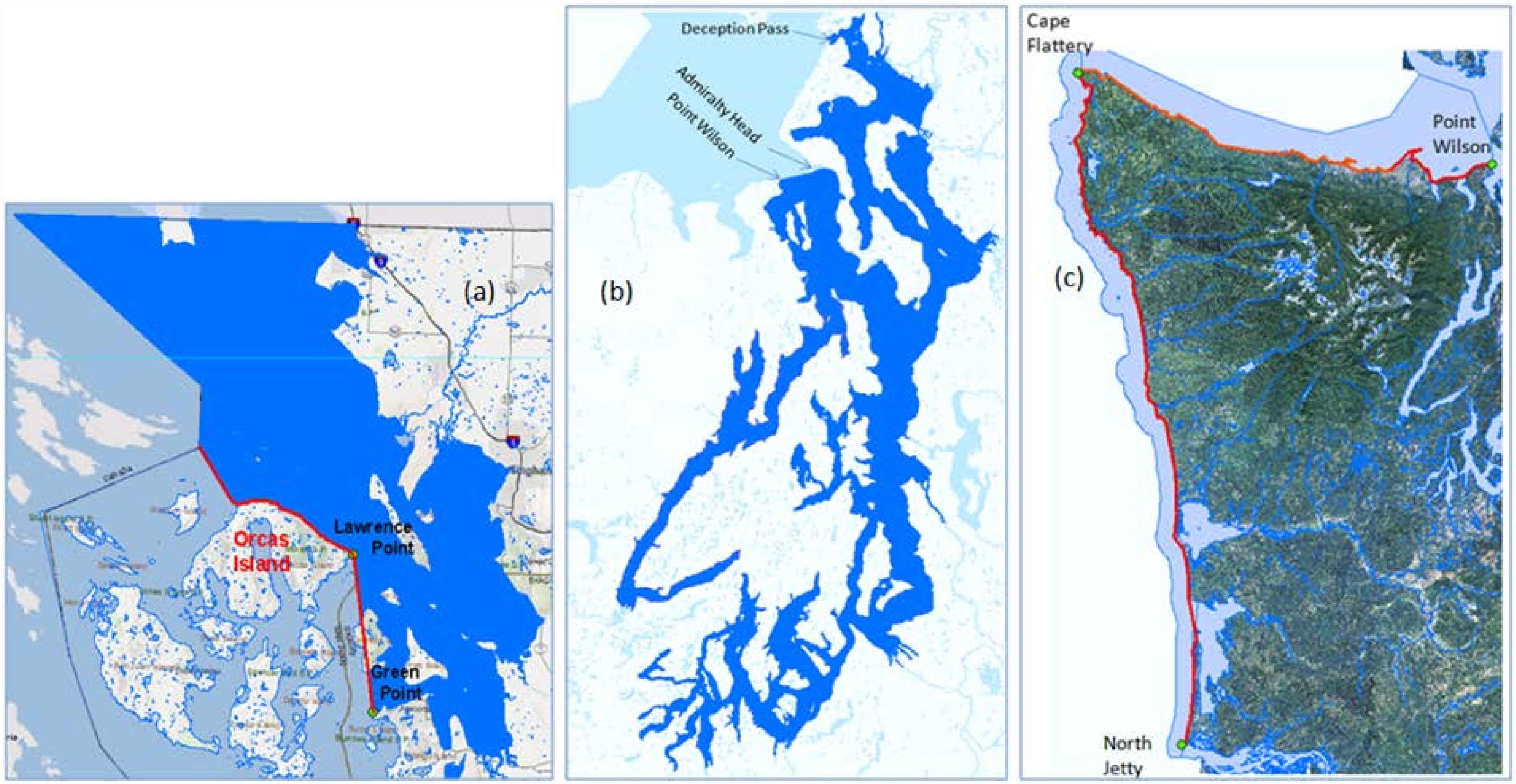


Figure C-14. (a) East of line through Green Point and Lawrence Point and area north of Orcas Island, (b) East of Deception Pass, and to the south and east of Admiralty Head, and south of Point Wilson, (c) waters existing within bays from Point Wilson westward to Cape Flattery and south to the North Jetty.

Attachment C.4 Using Continuous Simulation Dynamic Modeling to Establish Dilution Factors

Steady-state models maybe used to dynamically model discharge plume if data are available for each simulation time step. However, this is time intensive and requires extensive data including time-series for effluent and ambient flows, current velocities, densities, water depths and widths. The continuous simulation could be on an annual basis, seasonal or for a low flow month that is simulated for at least 15 years.

For freshwaters, if the discharge buoyancy is insignificant (both effluent and river are freshwaters and temperature differences are not significant), and the channel is wide enough (width >> 2 x depth) for the channel hydraulic radius equivalency to depth assumption (rectangular channel) to hold true, and for the discharge plume to be fully vertically mixed within a short distance, RiverPlume6 may be used to predict dilution. For continuous simulation using this model, the following procedure should be followed (see Figure C-15).

1) Continuous (daily) flow rate for both discharge and stream
2) Predict stream velocity, depth and width for a given flow at near the diffuser location
   a) using measured field data to generate "power equations" (Chapra 1997):
      i) Channel velocity, $U = aQ^b$
      ii) Channel depth, $H = \alpha Q^\beta$
      iii) Channel width, $B = cQ^f$

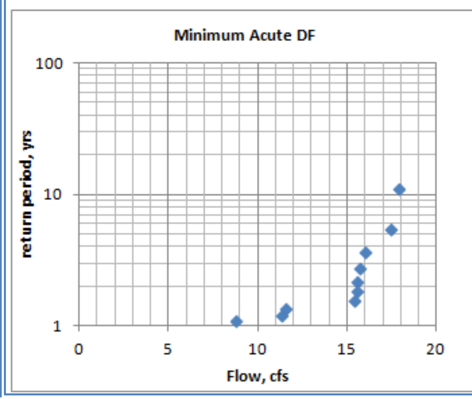
         where, Q = channel flow

         a, b, α, β, c, and d are empirical constants that are determined by stage-discharge rating curves for the channel cross-section near the diffuser. And, $b + \beta + f = 1$. For rectangular channel assumption the width will essentially remain the same regardless of the flow. The "power equations" may be established for different ranges of flows.
   b) Nearby USGS gage data on flow, velocity and water surface elevation.
   c) Using HEC-RAS model to generate site specific hydrological data relationships between flow, velocity, width and depth
3) Dilution from each discharge and ambient flow combination is compared with the maximum dilution allowed (based on flow) under law. Predicted plume width is also compared with maximum plume width allowed under law. The minimum dilution based on these considerations is then selected for the time-step.
4) A running 7-day average dilution factor time-series is generated. For each of the year of simulation, the minimum 7-day dilution factors are then tabulated.
5) A non-parametric method is then used to generate the dilution factor that has a recurrence interval of 10 years. EPA (1991) states that a hydrologically-based 7Q10 flow would be similar to biologically based 4-day 3-year design flows. If justifiable, a distribution based analysis (such as Log-Pearson III) could also be used.

	A	B	C	D	E	F
1	date	Min Acute DF	Min Chronic DF			
2	1-Mar-04	19.88	153.26			
3	2-Mar-04	20.42	159.06			
4	3-Mar-04	14.88	112.58			
5	4-Mar-04	18.35	118.32			
6	5-Mar-04	20.83	129.85			
7	6-Mar-04	24.39	152.26	year	7d avg Min	7d avg Min Chronic DF
8	7-Mar-04	25.32	161.69	2004	20.58	141.00
9	8-Mar-04	37.78	217.64	2004	23.14	150.20
10	9-Mar-04	51.78	278.41	2004	27.62	167.25
11	10-Mar-04	53.36	291.38	2004	33.12	192.79
12	11-Mar-04	41.97	245.44	2004	36.49	210.95
13	12-Mar-04	45.13	275.79	2004	39.96	231.80
14	13-Mar-04	36.16	229.18	2004	41.64	242.79
15	14-Mar-04	31.89	209.50	2004	42.58	249.62
16	15-Mar-04	30.63	204.63	2004	41.56	247.76
17	16-Mar-04	30.77	210.87	2004	38.56	238.11
18	17-Mar-04	32.72	221.63	2004	35.61	228.15
19	18-Mar-04	30.67	199.56	2004	34.00	221.59
20	19-Mar-04	32.61	213.05	2004	32.21	212.63
21	20-Mar-04	29.10	199.42	2004	31.20	208.38
22	21-Mar-04	26.93	188.70	2004	30.49	205.41
23	22-Mar-04	31.01	211.35	2004	30.54	206.37
24	23-Mar-04	40.87	258.83	2004	31.99	213.22
25	24-Mar-04	34.87	211.32	2004	32.29	211.75
26	25-Mar-04	29.91	190.22	2004	32.19	210.41
27	26-Mar-04	23.78	160.51	2004	30.92	202.91
28	27-Mar-04	17.54	122.03	2004	29.27	191.85
29	28-Mar-04	15.37	120.55	2004	27.62	182.12
30	29-Mar-04	13.87	110.79	2004	25.17	167.75
31	30-Mar-04	20.84	145.26	2004	22.31	151.53
32	31-Mar-04	25.72	174.03	2004	21.01	146.20
33	1-Apr-04	19.88	156.30	2004	19.57	141.35
3589	26-Dec-13	45.19	238.49	2013	41.59	204.52
3590	27-Dec-13	41.21	226.77	2013	43.77	213.13
3591	28-Dec-13	40.87	231.62	2013	44.95	223.45
3592	29-Dec-13	37.18	219.13	2013	45.10	230.31
3593	30-Dec-13	35.89	215.56	2013	44.14	234.42
3594	31-Dec-13	31.04	185.90	2013	40.58	225.27

Min DF for acute and chronic mixing zone boundaries are predicted by RiverPlume6 for each combination of ambient and discharge flow rates. A 7-day average minimum dilution factor is then extracted for each year to estimate the lowest 7-day average dilution factor with a recurrence interval of 10 years using a pivot table and non-parametric equations.

Row Labels	Values Min of 7d avg Min Acute DF	Min of 7d avg Min Chronic DF	return period acute, yrs	return period chronic, yrs
2004	15.32	123.29	1.57	11.00
2005	8.76	65.01	1.10	1.10
2006	15.70	107.91	2.75	2.20
2007	15.54	113.45	2.20	2.75
2008	11.26	85.23	1.22	1.38
2009	15.48	92.38	1.83	1.57
2010	15.96	115.59	3.67	3.67
2011	11.53	80.17	1.38	1.22
2012	17.39	102.66	5.50	1.83
2013	17.81	122.72	11.00	5.50



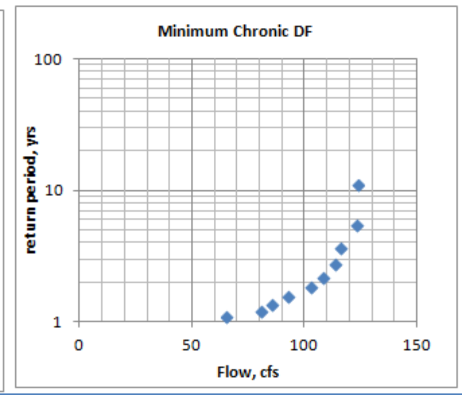


Figure C-15. Sample spreadsheet to run RiverPlume6 dynamically for daily inputs of discharge and ambient flows

Appendix D

1. Supporting Statistical Study for Performance-Based Reduction of Monitoring

(from EPA, April 1996)

1.1 Effect of Sample Size on Probability of Violation

EPA has done a statistical analysis on the effect of sampling frequency on compliance assessment. The basic premise underlying a performance-based reduction approach is that maintaining a low average discharge relative to the permit limit results in a low probability of the occurrence of a violation for a wide range of sampling frequencies.

The probability of the occurrence of a violation of a monthly average permit limit was calculated. Tables D-1, D-2, and D-3 display the percentage of time that a monthly average permit violation will be reported given sample size and a long-term average to permit ratio. This probability is dependent on the true long-term average of the discharge, the permit limit, and the monthly sampling frequency. The variables of long-term average and permit limit are both reflected in the tables by expressing these as a ratio. Tables D-1 through D-3 assume a normal distribution of monthly averages and show the effect of altering the assumed coefficient of variation, using 20%, 60%, and 80%, respectively.

Obviously, the best estimate of the true monthly average discharge is obtained by daily sampling. One can assess the true violation rate of a discharge by looking at the probability calculated assuming sampling was done daily (30 times per month). In order to maintain compliance with a permit limit, the long-term average level of the discharge must be controlled at a level less than the permit limit. Reducing the sample size, while increasing the probability that a violation will be reported, does not change the underlying probability of reporting a violation associated with a baseline estimate of the monthly average calculated with 30 samples. With a constant performance, the probabilities of reporting a permit violation increase as the sample size is reduced from daily sampling because the variance of the average is inversely proportional to the sample size.

Looking at the true violation rate of a facility sampling daily and operating at 75% of their permit limit, these tables show that the probability of a violation in a given month is 1% or less. If the long-term average discharge is 65% of the permit limit, the true percentage of violation is less than 1%. As sample size decreases for a given discharge/limit ratio, the expected percentage of time that the average of the samples collected during the month will exceed the permit limit increases. For example, Table D-3 demonstrates that at a ratio of 65%, the expected violation rate is effectively zero. If a subsample of 8 samples per month is taken instead of 30, the facility has a 3% chance of reporting a violation. If only one sample per month is taken, the chances of reporting a violation increase to 25%. The facility performance (true monthly average discharge) has not changed, thus "missed" monthly average violations are not an issue. The probabilities calculated for very low sampling frequencies reflects the risk assumed by the discharge operator that monthly average violations will be reported when in fact the process average is under permit limit. If facility performance degrades during the permit term and sampling has been reduced, it can be seen that the facility will have probability of reporting violations at a higher rate, even if the long-term average is still below the permit limit. An example will illustrate this point. Table D-3 shows that if a facility was judged to be at 75% of their permit limit and reduced sampling from 16 to 12 times per month, the probability of violation would change from approximately 5% to 7%. If the long-term average performance degraded to 90% of the permit limit, the 12 monthly samples would yield expected monthly average permit violations 32% of the time instead of 29% of the time if 16 samples were collected.

Table D-3 shows probabilities calculated using a more conservative assumption of 80% coefficient of variation. The results show that facilities with a long-term average of less than or equal to 75% have essentially no chance of violating a monthly average limit, hence facilities with this performance would be good candidates for performance-based monitoring reductions. The reductions in Table 34 were designed to maintain approximately the same level of reported violations as that experienced with their current (baseline) sampling.

Table D-1. Probability of Reporting Monthly Average Permit Violations at 20% Effluent Variability (CV = 0.20; Normal Distribution).

Monthly Sample Size

[1]LTA/Permit	30	28	24	20	16	12	8	4	2	1
100%	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**
95%	7%	8%	10%	12%	15%	18%	23%	30%	35%	40%
90%	0%	0%	0%	1%	1%	3%	6%	13%	22%	29%
85%	0%	0%	0%	0%	0%	0%	1%	4%	11%	19%
80%	0%	0%	0%	0%	0%	0%	0%	1%	4%	11%
75%	0%	0%	0%	0%	0%	0%	0%	0%	1%	5%
70%	0%	0%	0%	0%	0%	0%	0%	0%	0%	2%
65%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
60%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
55%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
50%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
40%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
30%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
20%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%

[1] Ratio of calculated average of at least 2 years of effluent data to monthly average permit limit.

Table D-2. Probability of Reporting Monthly Average Permit Violations at 60% Effluent Variability (CV = 0.60; Normal Distribution)

Monthly Sample Size

[1]LTA/Permit	30	28	24	20	16	12	8	4	2	1
100%	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**
95%	32%	32%	33%	35%	36%	38%	40%	43%	45%	47%
90%	16%	16%	18%	20%	23%	26%	30%	36%	40%	43%
85%	5%	6%	7%	9%	12%	15%	20%	28%	34%	38%
80%	1%	1%	2%	3%	5%	7%	12%	20%	28%	34%
75%	0%	0%	0%	1%	1%	3%	6%	13%	22%	29%
70%	0%	0%	0%	0%	0%	1%	2%	8%	16%	24%
65%	0%	0%	0%	0%	0%	0%	1%	4%	10%	18%
60%	0%	0%	0%	0%	0%	0%	0%	1%	6%	13%
55%	0%	0%	0%	0%	0%	0%	0%	0%	3%	9%
50%	0%	0%	0%	0%	0%	0%	0%	0%	1%	5%
40%	0%	0%	0%	0%	0%	0%	0%	0%	0%	1%
30%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
20%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%

[1] Ratio of calculated average of at least 2 years of effluent data to monthly average permit limit.

Table D-3. Probability of Reporting Monthly Average Permit Violations at 80% Effluent Variability (CV = 0.80; Normal Distribution)

Monthly Sample Size

[1]LTA/Permit	30	28	24	20	16	12	8	4	2	1
100%	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**	**50%**
95%	36%	36%	37%	38%	40%	41%	43%	45%	46%	47%
90%	22%	23%	25%	27%	29%	32%	35%	39%	42%	44%
85%	11%	12%	14%	16%	19%	22%	27%	33%	38%	41%
80%	4%	5%	6%	8%	11%	14%	19%	27%	33%	38%
75%	1%	1%	2%	3%	5%	7%	12%	20%	28%	34%
70%	0%	0%	0%	1%	2%	3%	6%	14%	22%	30%
65%	0%	0%	0%	0%	0%	1%	3%	9%	17%	25%
60%	0%	0%	0%	0%	0%	0%	1%	5%	12%	20%
55%	0%	0%	0%	0%	0%	0%	0%	2%	7%	15%
50%	0%	0%	0%	0%	0%	0%	0%	1%	4%	11%
40%	0%	0%	0%	0%	0%	0%	0%	0%	0%	3%
30%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
20%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%

[1] Ratio of calculated average of at least 2 years of effluent data to monthly average permit limit.

1.2 Detailed Protocol for Calculating Probability of Reporting Permit Violations

Calculation of probabilities for Tables D-1, D-2, and D-3

Probability distributions may be used to model effluent data and assess the probability of permit violations. The models provide a logical and consistent methodological framework for using observed performance data to assess permit limitations in an objective manner. The goal of the limitations is to establish performance levels that enforce good treatment and ensure that water quality objectives are met. In deriving limitations, sufficient allowance for variation in treatment performance is provided such that a well-operated treatment system should be capable of compliance with the limitations at all times. In using probability models as the basis for limits, it is necessary to select a percentile value such that, within the context of the model, any meaningful limit will have a non-zero probability of being exceeded.

The results shown in the tables here are derived from probability distribution functions that may be used to model effluent data. That is, the processes are assumed to operate over time in a manner that is consistent with past performance. No intervention to change the process or exert more or less control over the discharge is assumed.

Calculation of the probability that a reported permit violation will occur depends upon: the number of individual samples taken during the month, the long-term discharge level, the variance of the discharge concentrations, the probability distribution of the individual samples during the month, and the permit limit. There are two probability distributions commonly used to model effluent data: the lognormal distribution and the normal distribution. The lognormal distribution usually provides a good fit to data sets comprised of individual effluent measurements because such data typically have two critical lognormal characteristics: they are positive valued and positively skewed. Positive skewness means that the data are characterized by a tendency for a preponderance of measurements in the lower range of possible values with relatively fewer measurements stretched out over a wider range of possible upper values. The lognormal also has the property that the logarithms (natural or base 10) of the data are normally distributed. The normal distribution has the well-known "bell shape" and is mathematically straightforward so that working with the logarithms of effluent data is relatively uncomplicated.

The asymptotic distribution of sample averages is normally distributed. That is, the average of a sample of individual measurements will have a distribution that is approximately normally distributed regardless of the distribution of the individual measurements. The quality of the approximation depends on several factors including the number of individual measurements being averaged and the form of the underlying distribution. Although individual effluent measurements are rarely normally distributed, it is reasonable in many situations to approximate the distribution of the averages of effluent measurements with a normal distribution and thus the normal approximation is used in many cases as a model for monthly average effluent limitations. The results in Tables D-1 through D-3 are based on the assumption of a normal distribution for the *averages* of effluent measurements. Extensive discussion on the statistical modeling of effluent data and methodology for setting effluent limitations are contained in EPA's 1991 *Technical Support Document for Water Quality-based Toxics Control* (TSD).

The results of calculating probability of a reported violation of a monthly average permit limit are shown in Tables D-1 through D-3 under different conditions. The purpose of these tables is to provide some insight into the effects of changing monitoring requirements. The probability of exceeding the monthly limit when the long-term average of the discharge is at the desired value can be thought of as the Type I error rate (alpha-level) of the monitoring program.

When the long-term average exceeds the desired limit, the probability of exceeding the monthly limit is now the monitoring program's ability to detect violation increases if the long-term average increases over the desired level. It should be understood that if permit limits are held constant and performance measures such as long-term average discharge and variability of treatment do not change, then reducing the number of monitoring measurements used to calculate the monthly average causes the probability of a violation to increase for all values of the long-term average less than the monthly average permit limit.

This has a two-fold effect: (1) the chances of reporting a violation even when the long-term average is less than the desired level (the Type I error rate) go up (2) the sensitivity (ability to detect violations) of the program increases. The tables also show that if the average discharge level is held well below the monthly average limit, the chances of a violation are small. The three tables reflect three different levels of variation in the underlying daily data as measured by the coefficient of variation.

The coefficient of variation (CV) is the ratio of the standard deviation of the distribution to the mean and is often expressed as a percentage. The CV is a convenient measure for summarizing the relative variability in a data set. The results in Tables D-1 through D-3 use CVs of 20%, 60% and 80% respectively. A coefficient of variation of 60% was used in the TSD to describe a typical level of variation for lognormally distributed effluent data. CVs of 80% and 20% were used to show the effects of higher and lower levels of variability.

The probability distribution of the average of N daily measurements taken during a month, M_N, is given by the following normal probability density function:

$$g(M_N) = \frac{1}{\sqrt{\frac{2\pi}{N}}\,\sigma} e^{-\frac{N(M_N-\mu)^2}{2\sigma^2}}$$

where μ is the mean or long-term average, and σ is the standard deviation of the daily discharges. If μ_1 is the maximum monthly average allowed by the permit, then the probability that the monthly average exceeds the permit maximum is given by $P(M_N>\mu_1)$. Using simple algebra this probability can be rewritten as:

$$P(M_N > \mu_1) = P\left(\frac{M_N - \mu}{\frac{\sigma}{\sqrt{N}}} > \frac{\mu_1 - \mu}{\frac{\sigma}{\sqrt{N}}}\right) = 1 - \phi\left(\frac{\mu_1 - \mu}{\frac{\sigma}{\sqrt{N}}}\right),$$

where $\Phi(.)$ is the standard normal cumulative probability function (the Microsoft®Excel built-in function NORMDIST).

Since

$$\frac{\mu_1 - \mu}{\frac{\sigma}{\sqrt{N}}} = \frac{\sqrt{N}}{C}\left(\frac{1}{\left(\frac{\mu}{\mu_1}\right)} - 1\right),$$

where C is the coefficient of variation, then the probability of a monthly average exceeding the maximum allowable can be calculated using C, N, and the ratio of the long-term average to the maximum allowable monthly average using NORMDIST. This is how the values in Tables D-1 through D-3 were calculated.

Alternate approaches to probability calculations:

The probabilities in Tables D-1 through D-3 were calculated with the assumption that the distribution of the sample means is normal. Individual sample values are generally best fit to a lognormal distribution. As discussed in the TSD, the mean of small samples from a lognormal distribution is in most cases approximately lognormal. Probabilities can be calculated assuming a lognormal distribution by two different methods, a Monte Carlo technique and the Microsoft Excel built-in function LOGNORMDIST. The resulting probabilities will be very close to those in the normal distribution table for the sample sizes and discharge levels under consideration for monitoring reductions, although the probabilities calculated from these two distributions may not be comparable for all sample sizes and all discharge levels.

The statistical evaluations used in this analysis are intended for use only to illustrate the effect and benefits of this strategy, alternative statistical techniques and approaches may be utilized in other situations.

2. Determining the Number of Samples Required for Compliance Monitoring

This section provides some statistical tools for selection of a monitoring frequency when effluent data is available.

It is recommended that the permit writer use formula 3 with a confidence level of 90% and a relative error (d_r) around the mean of no larger than 20% (0.20) to derive a baseline monitoring frequency. Table D-4 is compiled from formula 3 for several combinations of confidence level, relative error and coefficients of variation. The rest of this section provides background material.

An effluent limit is a control parameter to assure that the long-term average (LTA) of a wastewater control device or practice is being maintained. The LTA may be derived from the performance of a wastewater treatment device or from a wasteload allocation necessary to meet water quality standards. We assume the number of samples required to demonstrate compliance is the same as that necessary to determine a mean of a sample from a population.

For a sampled population which is normally distributed, in which the samples are not correlated over time or space, and in which the number of elements in the sampled population (N) is large relative to the standard deviation (σ), the number of samples (n) required to estimate a population mean is:

$n = (Z_{1-\alpha/2}\,\sigma/d)^2$

where

$Z_{1-\alpha/2}$ is the standard normal deviate that cuts off $(100\alpha/2)\%$ of the tails of a standard normal distribution and

σ= standard deviation of the sampled population and

d = error around the mean.

This formula is for a two tailed test where the sample mean may be higher or lower than the true mean. In our situation we are only concerned with the situation in which the sample mean is higher than the true mean. Therefore, a one-tailed test is appropriate.

$$n = (Z_{1-\alpha}\, \sigma/d)^2 \qquad (1$$

(from Gilbert 1987)

To estimate a population mean with a error (d) of 20 (plus or minus 10) with a 10% probability of type I error (α)when the standard deviation (σ) of the population is 50 requires:

$$n = (Z_{0.10}\, 50/20)^2 = (1.2816*50/20)^2 = 10.2 \approx 10 \text{ samples.}$$

This assumes the data are independently, normally distributed and not correlated over time or space.

Effluent data are typically lognormally distributed (Schaeffer et al. 1980) but parametric statistics ($\overline{X}$ and s^2) are the best estimators of the population parameters if the coefficient of variation
(η) is less than 1.2 (Gilbert 1987). A typical coefficient of variation (η) for conventional pollutants from domestic wastewater treatment plants is 0.6. Other pollutants and industrial treatment processes typically have a higher CV of 1 to 1.5.

An adjustment for autocorrelation can be made to the formulas for sample numbers if necessary. This adjustment increases the number of samples required.

If the standard deviation is uncertain (because of limited previous sampling) then the *t* distribution should be used in the place of the standard normal distribution:

$$n = \left(t_{1-\alpha,\, n-1}\sigma / d\right)^2 \qquad (2$$

and the process of determining n becomes iterative.

Start with equation 1 above and use a Z value to approximate n,

$$n_1 = (1.2816*50/20)^2$$
$$n_1 \cong 10$$

then go to a t table to find $t_{0.90,\, 9}$ which is 1.383. Placing this value into equation 2 gives 12.

$$n = (1.383*50 \div 20)^2 = 11.95 \cong 12$$

Placing the t value for $t_{.90,\, 11}$ which is 1.363 into equation 2 gives 11.6 which is approximately 12 which then is the answer.

The margin of error can be expressed as relative error (d_r) of the mean instead of a absolute value and the coefficient of variation ($\eta = \sigma/\mu$) can be used as the measure of variability. The formula then becomes:

$$n = \left(Z_{1-\alpha}\eta / d_r\right)^2 \qquad (3$$

(from Gilbert 1987)

and a table can be produced of some common values (Table D-4).

Table D-4. Sample sizes required for Estimating the True Mean μ

Confidence level $(1-\alpha)$	Relative Error d_r	Coefficient of Variation (η)	
		0.60	**1.00**
.80 ($Z_{0.80}$ = .846)	.10	26	72
	.20	6	18
	.30	3	8
	.50	1	3
.85 ($Z_{0.85}$ =1.062)	.10	41	113
	.20	10	28
	.30	5	13
	.50	2	5
.90 ($Z_{0.90}$ = 1.282)	.10	59	164
	.20	15	41
	.30	7	18
	.50	2	7
.95 ($Z_{0.95}$ = 1.645)	.10	97	271
	.20	24	68
	.30	11	30
	.50	4	11
.99 ($Z_{0.99}$ =2.326)	.10	195	541
	.20	49	135
	.30	22	60
	.50	8	22

The permit writer may adjust sample sizes from Table D-4 according to factors discussed in Section 1.3.1 of Chapter 13.

Formula 3 can be adjusted for uncertainty of the standard deviation by use of the t distribution and an iterative process as in formula 2. The number of samples required for a given confidence level, relative error and cv will be slightly higher using the t distribution, however, not significant for the extra effort.

Zar (1996) presents an alternative method for determining n, the number of samples to determine the mean with specified type I (α) and type II (β) error probabilities, as;

$$n = \frac{s^2}{\delta^2}(t_{\alpha(1),\upsilon} + t_{\beta(1),\upsilon})^2 \qquad (4$$

where

s^2 is the sample variance estimated with v degrees of freedom

β is the probability of type II error

δ is the size of the error around the mean (d from above)

α, and t are defined as above.

In this equation the number of data required to calculate a confidence interval of a specified width depends upon: (1) the width desired - narrow widths require more samples, (2) the variability in the population - larger variability requires larger sample size, (3) the confidence level (type I error), and (4) the assurance that the confidence interval will be no larger than that specified (type II error).

This formula is also iterative.

To determine the number of samples required to estimate the mean of a sample with a 90% probability that the 95% confidence interval will be no wider than 3 mg/l (the effluent limit is 30 mg/l), then δ = 3 mg/l, β= 0.10, 1-α = 0.95, and α = 0.05. Assume an estimate of the population variance from previous sampling: s^2 = 18.0388 with v = 24. If we start with a guess of 8 samples per month then:

$t_{0.05(1),7}$ = 1.895 and $t_{0.10,(1),7}$ = 1.415

Using equation 4,

$$n = \frac{s^2}{\delta^2}(t_{\alpha(1),\upsilon} + t_{\beta(1),\upsilon})^2$$

n = (18.0388/9)(1.895+1.415)^2
n = 22

for n = 22, $t_{0.05(1),21}$ = 1.721 and $t_{0.10,(1),21}$ = 1.323

n = (18.0388/9)(1.721+1.323)^2
n = 19

for n = 19, $t_{0.05(1),18}$ = 1.734 and $t_{0.10(1), 18}$ = 1.330

n = (18.0388/9)(1.734+1.330)^2
n = 19

This formula gives a higher n because of the additional constraint of the type II error.

An Industrial Example

A permit manager proposes to use the following data to derive performance-based effluent limits for a discharge from a metal facility. The data are metal concentrations in mg/l from a settling basin. The permit writer wants to know the number of samples to determine the monthly average such that the type I error (false judgment of noncompliance) will be 5% with a 10% interval around the mean.

Mean	0.627260274
Standard Error	0.010826197
Median	0.59
Mode	0.5
Standard Deviation	0.206834198
Sample Variance	0.042780385
Kurtosis	5.752860027
Skewness	1.644693358
CV	0.33
Range	1.7
Minimum	0.29
Maximum	1.99
Sum	228.95
Count	365
Confidence Level(95.0%)	0.02128972

99th percentile = 0.2135155

a. Using formula 1 to determine the number of samples per month to determine a monthly average, with d = 0.06 (10% of the mean), and α =0.05. The data standard deviation s is used as an estimator of σ.

$$n = (Z_{1-\alpha/2}\,\sigma/d)^2$$

$$n = (1.645 * 0.2068 / 0.06)^2 = 32 \text{ samples/month}$$

b. Using formula 2, d = 0.06 or 10% of the mean, α =0.05 and, the data standard deviation, s, as an estimator of σ.

$$n = \left(t_{1-\alpha/2,\, n-1} \sigma / d \right)^2$$

The Z value is used as an initial estimator for t.

n = ($Z_{.95}$* 0.2068 / 0.06)^2 = (1.645 * 0.2068 / 0.06)^2 = 32

n = ($t_{.95,31}$ * 0.2068 / 0.06)^2 = (1.696 * 0.2068 / 0.06)^2 = 34

n = ($t_{.95,33}$ * 0.2068 / 0.06)^2 = (1.693 * 0.2068 / 0.06)^2 = 34 samples/month.

c. Using formula 3 with the coefficient of variation (η) = 0.33, relative error (d_r) = 0.10 and α = 0.05.

$$n = \left(Z_{1-\alpha} \eta / d_r \right)^2$$

n = (1.645 * 0.33 / 0.10)^2 = 29 samples/month

Using t values in this formula will increase n slightly to 31.

d. Using formula 4 to determine n with β = 0.1 (probability noncompliance is occurring but not detected).

$$n = \frac{s^2}{\delta^2} (t_{\alpha(1),\upsilon} + t_{\beta(1),\upsilon})^2$$

$$n = \frac{0.0428}{(0.0627)^2} (1.645 + 1.282)^2$$

$$n = 93$$

Appendix E

1. What is a Total Maximum Daily Load?

The following is a general, simplified summary that describes how and why Ecology establishes a Total Maximum Daily Load (TMDL). Links are provided below for more detailed information.

The federal Clean Water Act establishes a two-pronged attack on pollution:

1. Pollution point sources have to meet a technology-based permit limit under their NPDES permit.
2. A water quality-based system is set up to address fresh and marine waters that are still impaired by pollution, even after NPDES technology-based permit limits are in place and fully implemented.

The water quality-based system consists of the following steps:

1. Water quality standards are adopted under a rule that defines the designated uses that depend on water quality and the criteria necessary to meet those uses.
2. Effluent limits in individual NPDES permits must ensure that the discharge will not contribute to a violation of applicable criteria. This issue is described in detail in Chapter 6 of the *Permit Writer's Manual*.
3. Water bodies are monitored and data from monitoring collected, assessed for data quality, and compared to the water quality standards to determine compliance.
4. Water bodies evaluated as part of Ecology's Water Quality Assessment and not meeting water quality standards are included in a list of impaired waters in Category 5. This is also called the "303(d) list", from the section of the Clean Water Act that includes this requirement (https://ecology.wa.gov/Water-Shorelines/Water-quality/Water-improvement/Assessment-of-state-waters-303d).
5. The CWA requires that impaired waters be studied to determine the cause of the impairment and to calculate the "total daily maximum load" of pollutants that can be discharged into the water body and still allow state standards to be met. This TMDL study looks at the cumulative effects of natural conditions, nonpoint sources, and point sources. In compliance with an MOA with EPA, Ecology schedules these TMDL studies based on agency priorities and available resources.
6. Based on the TMDL study, Ecology allocates the pollution sources into *wasteload allocations* (WLA) for NPDES-permitted point sources, and *load allocations* (LA) for nonpoint, diffuse, or other non-permitted sources. Load allocations can also account for natural background conditions. *A reserv*e for future growth may be provided through a specified *WLA or LA*. A margin of safety must be identified to account for uncertainty, which can either be implicit (based on conservative assumptions, for example) or explicit (provided as a load allocation). The TMDL is the sum of all these allocations, and is sometimes described as:

$$\begin{aligned} TMDL = &\Sigma\ WLAs \\ &+ \Sigma\ LAs \\ &+ LA \text{ for natural background} \\ &+ \text{future growth LA and/or WLA} \\ &+ MOS \end{aligned}$$

7. For each permitted point source that receives a WLA, the permitting agency then has to translate that WLA into permit conditions.

Many of these steps require the oversight or approval of the U.S. Environmental Protection Agency (EPA).

Note that the term “TMDL” is often applied to the TMDL study, Ecology’s process to establish TMDLs, and to the submittal package sent to EPA.

The TMDL program is complex, and on-line resources cover the subject in more detail.

- A detailed description of the TMDL program from EPA’s perspective can be found here: http://water.epa.gov/lawsregs/lawsguidance/cwa/tmdl/index.cfm.
- EPA has developed training modules, which can be found on the website “Web-based Total Maximum Daily Load (TMDL) and National Pollutant Discharge Elimination System (NPDES) Permit Training” (EPA, 2009; http://water.epa.gov/polwaste/npdes/Web-based-Total-Maximum-Daily-Load-TMDL-and-National-Pollutant-Discharge-Elimination-System-NPDES-Permit-Training.cfm).
- EPA provided some guidelines for TMDLs and NPDES permits in 2013 (EPA, 2013).
- Ecology’s website for TMDLs begins here: https://ecology.wa.gov/Water-Shorelines/Water-quality/Water-improvement/Total-Maximum-Daily-Load-process.
- A set of guidelines for TMDL modeling and analysis were published in the 1990s (Ecology, 1996).

2. The TMDL Process

Viewed from the perspective of an NPDES permit manager, the steps described above that are part of the process leading up to the establishment of a TMDL will now be explored in more detail.

2.1 Water Quality Standards Review

The starting point for any TMDL is the water quality standards. Permit managers should review the sections of the standards that apply to the receiving water both at the point of discharge and downstream of the discharge for permits they manage. Sometimes different criteria or other requirements may apply to downstream segments that must be considered, since the discharge may affect those segments.

This review should include the designated uses to be protected, criteria for those uses, any special conditions for that water body, and anti-degradation provisions. All water bodies have designated uses and associated criteria even if they are not specifically listed in Table 602 of the standards. One tool to quickly determine the proper beneficial uses and water quality criteria for a waterbody is the Water Quality Atlas (in the Atlas, choose the "i" button, click on the waterbody, then choose the WQ Standards tab). For "default" uses and criteria, refer to WAC 173-201A-600.

Note that the standards often have special conditions that apply solely to NPDES permit discharges. Be sure to read table 602 closely and/or read all the information provided by the Water Quality Atlas.

TMDLs usually look at the effect of wastewater discharges outside of the mixing zone (far-field impacts – dissolved oxygen is an example). For that reason, it is good for permit managers to know the water quality criteria downstream of a discharge where far-field effects may occur. Detailed information on how the Water Quality Standards protect state waters is available here: https://ecology.wa.gov/Water-Shorelines/Water-quality/Freshwater/Surface-water-quality-standards.

On a regular basis Ecology's Water Quality Program (WQP) conducts a review of the standards (sometimes called the "triennial review" although it usually occurs on time frames other than every three years). Changes proposed in this periodic review may affect a permit by restricting the ability to change NPDES permit conditions or by its effect on a TMDL (that would also result in limitations on a permit). Therefore, permit managers may want to comment on potential changes regarding their impact on permitted facilities.

A few points about water quality standards and TMDLs are important for the permit manager to be aware of:

- A TMDL is based on the standards that exist at the time of TMDL development. TMDLs may show that the water quality of a waterbody is higher than specified in the standards, or that natural conditions do not meet water quality criteria, or that more stringent water quality is needed in upstream areas to protect downstream uses. However, TMDLs do not change the standards. A TMDL study could suggest a change the standards, but historically that has not occurred.
- There are other processes by which Ecology may propose changes to the standards. These include establishing "site-specific standards" and conducting a "Use Attainability Analysis" or UAA. These processes are complex and controversial and are rarely used. A TMDL has no direct connection to these processes, other than possibly triggering a discussion about changing the standards.
- The water quality standards refer to "natural conditions," defined as "surface water quality that was present before any human-caused pollution." The standards recognize that natural conditions may not meet the numeric criteria. Usually small alterations to natural conditions are allowed, eg. an increase in temperature of 0.3 degrees Celsius above natural conditions.

Field studies and modeling done in TMDLs sometimes determines a waterbody's "system potential" or "natural background" under critical conditions. The system potential or natural background is typically our best current approximation of natural conditions under a worse-case scenario. Given the complexity of the ecological changes caused by human activities over hundreds of years, the true nature of natural conditions is beyond the scope of any study. Nonetheless, a TMDL often evaluates natural conditions, usually through analysis of system potential or natural background conditions, which may form the basis of TMDL allocations.

One example of the challenges this issue poses: Natural conditions effectively refer to conditions that existed "before pioneer settlement." However, channel alterations or diverted water flows (such as in irrigated areas) may have completely changed a stream from its pre-settlement conditions. In these cases the TMDL may need to evaluate natural conditions as the potential conditions that would exist in the system absent human-caused pollution, but given the existing channel and flow alterations.

This is an area of controversy, so the TMDL team should consult with the Water Quality Planning Unit and in some cases include the Water Quality Management Unit, preferably at the time of the TMDL QAPP development. Some guidance is available, but this is an area of evolving policy. Nonetheless, although a TMDL cannot regulate channel or flow alterations, the TMDL study might identify improvements to those conditions that should be implemented. These improvements could minimize the impact of pollutants or better protect the supported uses.

2.2 Water Quality Assessment

Section 303(d) of the Clean Water Act requires the state to assess the quality of its waters and compare them to the water quality standards as currently adopted. Water bodies that are not meeting standards due to the discharge of pollutants (from either point or nonpoint sources or both) are included in the state's 303(d) list of impaired waters. More details can be found here: https://ecology.wa.gov/Water-Shorelines/Water-quality/Water-improvement/Assessment-of-state-waters-303d.

Ecology's Water Quality Program conducts an "integrated assessment", where water quality data are compared to the standards. Based on that comparison, a water body can be placed into one of seven categories. A detailed description of the categories is here: https://ecology.wa.gov/Water-Shorelines/Water-quality/Water-improvement/Assessment-of-state-waters-303d/Assessment-categories.

A few points about the categories are helpful:

- The 303(d) list is called "Category 5" and it is effectively the "TMDL To-do List".
- If a water body is not meeting standards, but there is sufficient evidence that it's solely due to a natural condition, it is not included on the 303(d) list. But if there are potential human impacts combined with natural effects, it does go on the 303(d) list.
- If a water body already has an EPA-approved TMDL in place, it goes into Category 4A.

- If a water body is impaired but there is a program being implemented in the water body that Ecology expects will result in compliance with the water quality standards, the water body may be placed in Category 4B. For example, this could occur if a single NPDES permit can be shown to address the source of impairment or if a program such as Ecology's Eastern Region's Livestock and Water Quality Program is being implemented in the water body and will address the impairment.
- If a water body is impaired by "pollution" but not by a "pollutant" – for example, problems with flow or habitat loss – then it would be listed in Category 4C.

When the WQP is updating the assessment, the bottom line for a permit manager is this: , review the listings for any water bodies to which your facilities discharge. It will help the permit manager to be aware of proposed listings that could affect the facilities for which he or she is responsible. In addition, the expertise of the permit manager could help to ensure that the assessment places the water body in the correct category. The best way to do this is to use the draft WQ Assessment Map that will be available during the review period.

2.3 TMDL Selection, Management, and Outreach

2.3.1 TMDL Selection

There are several different ways a TMDL project can be identified, prioritized, and selected for a TMDL study.

Proposals for TMDLs generally start in the regional offices. WQ TMDL Leads may consult with major stakeholders and/or participate in a program-wide "soiree" to propose new TMDL projects. TMDL Leads identify permits located in proposed TMDL areas and should consult with permit managers. This is also an opportunity for regional Permit Managers to work with their regional TMDL Leads to propose a TMDL study that can help address permit management challenges resulting from a 303(d) listing. Most TMDL studies are conducted by the Environmental Assessment Program (EAP), but some may include local partners or be conducted directly by WQ program. Each year in the winter, regional offices prioritize their requests.

Staff from the WQP and EAP confer through late winter and spring of each year to evaluate the proposed projects by WQP priorities and EAP resources, as well as by timing, scope, and feasibility. Projects are then tentatively classified as "will do", "conditional", or "will not do". Permit managers should work closely with the TMDL regional leads to track upcoming TMDL studies that will affect the NPDES permits they are responsible for.

TMDL projects may also arise for other reasons or be initiated by entities other than EAP.

- At times EPA may take on a TMDL, especially if it involves multiple states and is controversial. An example is the dioxin TMDL developed for pulp mills in Washington and Oregon.

- A TMDL can be initiated by the availability of dedicated funds from EPAgrants or technical support resources for a TMDL study that can be matched to a particular TMDL need.
- Local governments may initiate a TMDL study to address a 303(d) listing that is affecting their jurisdiction.
- Because of the recent "Pinto Creek" decision[1], existing or prospective NPDES permittees may request a TMDL study in order to obtain a WLA or compliance schedule that would allow for a new discharge into an impaired water segment..

Generally EAP will play a technical support or co-management role in TMDL studies proposed by third parties. However, there are a few cases where Regional WQP staff may conduct or manage the TMDL study directly.

2.3.2 TMDL Roles and Organization

Once a TMDL project is selected, there are a variety of roles for Ecology staff:

- A TMDL regional lead is identified in the Water Quality Regional Section (or field office, such as the Bellingham Field Office). The regional lead is responsible for shepherding the project from beginning to end and being the primary point of contact both internally and with EPA and external stakeholders.
- The TMDL technical lead, usually from EAP, will organize and conduct the TMDL study, and provide oversight to any contractors or other technical staff participating in the study. They provide technical support to the TMDL regional lead.
- Specialized technical staff may be involved from EAP or consulting firms. This could include field survey specialists, a hydrogeologist to conduct a ground water study, flow measurement staff, or modeling specialists.
- Permit managers with permits affected by the TMDL need to be engaged in the TMDL development process. Their involvement is critical to provide information on the discharger and to coordinate interactions with the permittees in consultation with the regional lead. This also includes the managers of general permits, who may work out of headquarters. The regional TMDL Lead is responsible for keeping permit managers up to date on TMDL progress and for facilitating good communication between EAP modelers and permit managers.
- Other WQP specialists may become involved, such as nonpoint source inspectors, forest practices staff, or stormwater specialists.
- EPA designates a staff to oversee each TMDL in Washington State. In TMDLs that are expected to be complex or controversial, or which cross jurisdictions with other states or Tribal lands, the EPA representative will likely take an increased TMDL role for providing support and guidance.

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[1] http://cdn.ca9.uscourts.gov/datastore/opinions/2007/10/03/0570785.pdf

- Tribal reservation lands (i.e., Indian Country) are outside Washington State jurisdiction and are managed by the Tribes with EPA oversight. If a TMDL includes reservation lands, Tribes will decide if they want to participate in a partnership approach, let the state take the lead but provide input, or exclude themselves from the TMDL. In the former cases, the affected Tribes may designate a lead to work with Ecology and EPA staff.
- In some cases a TMDL may be conducted for a water body shared by two states. If a bi-state approach is pursued, the neighboring state will have staff involved with the TMDL, along with EPA coordination.

2.3.3 TMDL Public Outreach

Early in the process, the TMDL technical lead and regional lead (staff from the Water Quality Regional Section or Field Office) should begin working on an outreach strategy. There are several steps that are common in most TMDL projects:

- Organize a local stakeholder group to be the focus of report review and participate in reviewing allocations and developing the implementation strategy.
- Identify representatives of various levels of government who should be involved, depending on the jurisdictions affected by the TMDL. These may include:
    - Federal agencies, such as the U.S. Forest Service or Army Corps of Engineers.
    - Other State agencies, such as Fish and Wildlife or Health.
    - Tribal agencies, where the watershed for the TMDL includes reservation lands or usual and accustomed Tribal fishing areas.
    - Local government and municipal corporations, such as counties, cities, Ports, or PUDs.
    - Nonprofit groups, citizens or business groups significantly affected by the TMDL or involved in TMDL implementation.
- Obtain input on the Quality Assurance Project Plan (QAPP) through public comment, advisory group, or a technical subset of an advisory group.
- Conduct a public review and comment period for the draft TMDL technical study report and the final TMDL if published separately, or for the combined technical study and TMDL.
- Sponsor periodic meetings of the stakeholder group to update them on progress of the study, discuss complex or controversial elements of the study, and get feedback and hopefully buy-in to the overall scope, process, and planned endpoint of the TMDL project.

Each permit manager has a key role of working with the regional TMDL Lead to bring permittees into the process, helping them to understand the TMDL, and working with them and the Ecology TMDL team to develop appropriate, effective, and reasonable implementation.

2.4 The TMDL Study

2.4.1 Study scoping

For an EAP-led TMDL study, once a potential TMDL has been selected EAP staff will work with WQP staff to develop an "agreed-upon scope of work." This is a summary scoping and an opportunity to further refine the approach of the TMDL study. There are a variety of technical approaches to a TMDL study, but they require varying levels of time and resources. Based on staff knowledge and experience, the rough outline of the study will be described. This can include the availability of third-party data, an EAP monitoring and sampling study, and analytical tools such as statistical analysis and modeling.

Although TMDL Leads should have already identified permits within a proposed TMDL footprint, this scoping should identify all NPDES dischargers within the TMDL footprint (discharging into the streams being studied and upstream in the watershed) and discuss them and their involvement in the study in greater detail.

The TMDL technical lead and permit manager should coordinate to determine how to get effluent data for the study. Although most DMR data as well as permit fact sheets are available through PARIS, it is helpful to supply the EAP TMDL technical lead with any other data (mixing zone studies, etc.) or other specific information of importance that could affect study design. It will help the permit manager to begin planning for possible effluent sampling and for the interactions with the permittee that are likely to occur during the course of the study.

Depending on the TMDL parameter under study, NPDES permits may be identified as minor sources. Although monitoring may not be needed for these discharges, they will ultimately still need to be included in the TMDL.

2.4.2 The Quality Assurance Project Plan

When study scoping is complete and the study formally starts, the first step will be the development of a Quality Assurance Project Plan (QAPP) for the study. The EAP TMDL technical lead will develop this document for an EAP study. For third-party studies or EPA contractors, the QAPP will likely be developed by the technical lead for those entities. It is not necessary for a permit manager to become fully familiarized with the elements of the QAPP. For permit managers wanting to learn more, the following information sources can be consulted.

A QAPP is required by agency policy, and the use of credible data in TMDL development is a requirement of state law. More information on data Quality Assurance (QA) is available here: https://ecology.wa.gov/About-us/How-we-operate/Scientific-services/Quality-assurance.

EAP has developed a template for a TMDL QAPP, which can be found here: QAPP Template - TMDL. The template lays out the structure for the QAPP and the issues it should address. Table E.1 shows an outline for the QAPP template.

A detailed description of the requirements of a QAPP is presented in Ecology's publication, *Guidelines for Preparing Quality Assurance Project Plans for Environmental Studies*, which can be found here: https://fortress.wa.gov/ecy/publications/summarypages/0403030.html.

Table E.1 Quality Assurance Project Plan (QAPP) Outline

QAPP Section and Subsection Headings	Description
What is a Total Maximum Daily Load (TMDL)?	
o Federal Clean Water Act requirements	Summary of Section 303(d)
o TMDL process overview	Brief summary of process context from 303(d) list to a final TMDL implementation plan.
o Who should participate in this TMDL?	A map of the project footprint (the geographic extent of the study) and descriptions of stakeholders, including major entities holding NPDES permits.
o Elements the Clean Water Act requires in a TMDL	Definitions of key terms, including: Loading Capacity, Allocations, Seasonal Variation, Margin of Safety, Reserve Capacity, and Surrogate Measures.
Why is Ecology Conducting a TMDL Study in This Watershed?	
o Background	The history that led to the selection of this TMDL study.
o Study area	Description and map of the study area (with more details than the previous map, such as impaired reaches and land use covers).
o Impairments addressed by this TMDL	Description of the 303(d) listings that triggered the TMDL.
o How will the results of this study be used?	A short narrative about how the study leads to a TMDL and implementation activities.
Water Quality Standards and Numeric Targets	
o Parameters	A summary of the Water Quality Standards for each parameter being addressed by the TMDL.
o Global Climate Change	A short narrative about the TMDL in the context of climate change.
Watershed Description	A detailed overview of the study area that can include subsections on: geographic setting, geology, climate, hydrology and streamflow, wildlife, vegetation, channel alterations, land uses, permitted discharges, and other stakeholders and affected jurisdictions.
Historical Data Review	A description of the spatial and temporal patterns of historical environmental data relevant to the study.
Goals and Objectives	Project goals and the study objectives to meet those goals.
Study Design	
o Overview	A summary of the approach taken for the TMDL study.
o Modeling and Analysis Framework	A description of the analytical framework for the study, including modeling or statistical tools, the data requirements of the tools, and how they affect the field study design.

QAPP Section and Subsection Headings	Description
o Details	A description of the planned field monitoring and sampling program, including fixed network sampling, special studies, and storm monitoring. Also, any other relevant activities, methodologies, or information sources to be used in the study to meet study objectives.
Sampling Procedures	Specific procedures for collecting samples.
Measurement Procedures	Specific procedures for field and laboratory measurements.
Quality Objectives	A summary of the methods to assess the quality of information used in the study.
o Measurement Quality Objectives	Descriptions of MQOs for the laboratory and field data being used by Ecology for this study. Definitions for precision, bias, and reporting limits for data. Descriptions of the methods that will be used to assess those objectives (such as %RSD or RMSE). A table for the specific values of the objectives.
o Quality Objectives for Modeling or other AnalysisAnalysis	A description of the quality objectives for the data analysis. For modeling, this will be the precision and bias objectives for model calibration and verification and any other “goodness-of-fit” methods being selected. For statistical analysis, this might be the target r2 or P value for the statistics used. Includes definitions, methods, and a table of target values for the quality objectives.
Quality Control	Procedures to meet QA objectives, such as calibration, replicates.
Data Management Procedures	Procedures for recording and archiving data.
Audits and Reports	Procedures for data quality review and reporting.
Data Verification and Validation	Technical procedures for ensuring data quality.
Data Quality (Usability Assessment)	Summary statement on how data will be used based on QA results. Should address new data, external and historical data, and modeling or other analytical results.
Project Organization	Description of staff with responsibilities for the project. A table of names and contacts corresponds with the front signature page. Other staff with roles can also be explained narratively.
Project Schedule	A table of key dates for field and laboratory work; data archiving in Ecology’s Environmental Information Management system (EIM); and interim, draft and final reports.
Laboratory Budget	A table of parameters, dates, and costs.
References	Technical and regulatory references for the QAPP.

QAPP Section and Subsection Headings	Description
Appendices	Glossary, Acronyms, Abbreviations, and any other appropriate topics.

Some key points regarding QAPPs of relevance to permit managers:

- The QAPP is the guiding document for the TMDL study. By signing the QAPP, staff are accepting their roles and responsibilities in the project.
- The QAPP goes through a variety of reviews, including supervisors, clients, and technical consistency and peer reviews. Sampling should not begin until reviews are complete and the QAPP is approved by the technical lead's supervisor.
- The TMDL regional lead usually conducts the "client review" for a TMDL study QAPP in the TMDL project area. This is the opportunity for TMDL leads to communicate with permit managers, to ensure that the study appropriately addresses all affected permitted facilities. Providing review and input is valuable to:
    - Ensure that information in the QAPP is accurate.
    - Understand activities that will occur at the facility (such as effluent sampling) and that will affect the facility.
    - Advocate for study elements that will be useful for future permit management needs.
- An approved QAPP "locks in" the scope and schedule of the project.
    - If emerging conditions or information result in a need for significant changes to the scope of the project plan, a QAPP addendum must be written and approved by all the signatories of the original QAPP.
    - Simple changes in schedule are approved through a *Report Change Form*.
    - Minor adjustments to project activities due to conditions found in the field or the results of analysis are normal and don't require formal approval. However, periodic reports from the technical lead to the client are encouraged to ensure adequate communication about the progress and evolution of the project.
- The QAPP is a road map. The specific outcome of the study is usually uncertain and sometimes surprising. Adequate communication with permittees is vital to help prepare for a constructive evaluation of TMDL implementation alternatives when the project is complete.

2.4.3 The TMDL Study

In this section the overall approach to TMDL studies will be described. This is provided as a summary description. Each study is unique, and there are always a multitude of details that will not be provided here.

2.4.3.1 General Considerations – Field Study

In most cases the TMDL study consists of three phases: the field study; the TMDL analysis; and drafting of the TMDL; and stakeholder outreach with public review. In most cases EAP collects all field data and performs the TMDL analysis.. In a few cases a TMDL uses existing data or data collected by an external party such as a wastewater treatment plant. Sometimes a consultant (such as one provided by EPA) does the TMDL analysis. In all these cases, an Ecology technical lead still coordinates the study.

The field study phase should follow the outline of the QAPP. It's likely that specific details and logistics will be worked out during the study and that conditions in the field might trigger some adjustments.

For the permit manager, there are some opportunities and needs for engagement during the field study:

- The study may include sampling of a permitted discharger's effluent. The technical lead may need the permit manager's help with understanding the nature of the waste stream and treatment processes and with scheduling visits and entering the facility. When a regional TMDL Lead becomes aware that an upcoming TMDL will require atypical WWTP data (e.g., phosphorus monitoring) they may inquire about the possibility of including additional monitoring requirements in a permit.
- The study may need to obtain detailed records from the facility, such as continuous flow records or complete sampling results. Again, the permit manager can help and will want to be involved with this.
- The permit manager may wish to offer help on a field survey. It's a good way to get to know the technical and regional leads, learn more about the study, and see the permitted discharger in a broader environmental context.

2.4.3.2 General Considerations – TMDL Analysis

When the field work is completed, the next step is analysis of data. The data must be reviewed and the quality analysis completed. Data that have been quality-checked are then loaded into EIM. At this point the full-blown analysis can begin.

During the TMDL analysis process, help from the permit manager may again be needed. The technical lead may need specific information about a discharger for modeling or other analysis, such as:

- Location of the outfall.
- The dilution ratio that applies at the edge of the mixing zone.
- Effluent characteristics, such as design flows, seasonal performance-based flows, or effluent monitoring results.

As the TMDL analysis is nearing completion, support from the permit manager becomes particularly critical. The TMDL analysis generally follows three steps:

- The first step of TMDL analysis attempts to reproduce the observed conditions, usually by setting up a model and calibrating it with field data. The permit manager can help with model calibration inputs, especially effluent discharge characteristics, but also receiving water characteristics, meteorological information, and other local factors.

- In the second step the calibrated model is used to estimate a waterbody's "system potential" or "natural background" conditions, depending on the parameter being evaluated. These conditions represent the local water quality wwith human influences removed to the extent possible given the modeling tools, project scope, and regulatory context. At a minimum, identified point sources and human nonpoint sources are removed from the model.

  Commonly, natural flow conditions are not estimated. However, if an analysis of flow impacts is needed, such as for dam releases, the effects of flow may be analyzed separately from the effects of pollutants. And depending on the modeling platform, the flow from a discharge may be left in the model to maintain the flow balance, but the effluent will be set to natural background pollutant concentrations so in the model it resembles a natural creek. This step is usually needed for temperature or dissolved oxygen (DO) TMDLs. It may not be needed for bacteria or toxics TMDLs, depending on the site-specific situation. Again, the local knowledge of the permit manager could be helpful with this step, especially given that the loading assigned to natural background can impact the capacity left for permitted discharges.

- The third step in the TMDL analysis is to determine the "Loading Capacity". Here human pollution sources are reintroduced into the model to determine the maximum amount of pollutant that the waterbody can receive and still meet water quality standards. This scenario generally is based on critical conditions, which include critical flows and meteorological conditions, and background water quality conditions. For TMDLs where low flow is critical, guidance on choosing low flows for modeling can be found in Guideline #2 of Ecology (1996). However, for some parameters, critical conditions may be average conditions or storm events. This is discussed in more detail below for each parameter.

  The assistance of the permit manager is critical in this step. Based on the calibration and natural conditions analysis, it may be clear that point sources will need to reduce their loading. This may include both individual permittees and dischargers covered by a general permit. Although wasteload allocations (WLAs) are not fully developed at this point in the TMDL process, some likely scenarios need to be developed for effluent pollutant loading for permitted dischargers. And it is possible that these initial estimates will continue to be used long into the project. Therefore, help from the permit manager is critical to determine pollutant limits for the Loading Capacity analysis that have some basis in potential implementation strategies.

  As discussed later, the EAP technical lead, regional TMDL lead, and the permit manager propose final WLAs to WQ managers as part of the draft TMDL review. Any potential WLAs with significant policy implications should be discussed with managers as early in TMDL development as possible.

There are several approaches to determining levels to set effluent loading from NPDES discharges:

- Apply design flows and permit limits. Depending on the analysis, this may apply to the conditions during the days being modeled, to seasonal conditions, or to monthly or semimonthly conditions. Guidance on choosing effluent flows for modeling can be found in Guideline #3 of Ecology (1996).

- Determine performance-based limits, especially if there are no permit limits or if actual performance is lower than permit limits. The TMDL technical lead can put performance-based limits into the model to see if they are adequate to protect water quality standards, in combination with some reasonable nonpoint source controls.
- Evaluate some logical additional treatment steps for the permittee. Two examples are: changing to UV disinfection for bacteria and applying advanced filtration for phosphorus removal.
- Determine effluent loading based on meeting standards "end-of-pipe" or at the edge of a mixing zone.
- Determine whether a no-discharge requirement is an option. This could be based on effluent recycling, ground water discharge, application to a crop or landscaping, or other strategies to treat and reuse effluent.
- Consider combinations of these approaches applied seasonally or set by some other criterion such as flow or treatment performance.

Unless treatment technology is based on "end-of-pipe" or technology-based limits, effluent loading used in modeling scenarios (and potential WLAs) will likely rely on dilution factors (DFs) to be met at the end of a mixing zone. If the TMDL uses DFs different from the DFs set in the NPDES permit, there could be some confusion and a need for additional work to harmonize and justify the differences. Ideally, the mixing zone approaches used for the TMDL and future permits should be the same. The permit manager should work with the TMDL technical lead to ensure this occurs.

Instead of evaluating and applying performance- or treatment-based loading, the TMDL technical lead might apply across-the-board percent reductions. Although this is simpler in the short run, it may result in pollutant loadings that are not realistic or cost-effective. This could complicate things or create more work in the long run. Maintaining good communication between the TMDL technical lead and the permit writer is essential to reduce confusion and keep both permit development and TMDL development on schedule.

The Regional TMDL lead and regional permit writer should engage the permitted entity early in the TMDL development process to educate them about the TMDL and provide an opportunity to explore potential future TMDL implementation. Permit managers and regional TMDL Leads must use their judgment on the best strategy to engage their permittees. It is important to make sure permittees understand that TMDLs for other parameters may be developed separately in the future. Any treatment or facility upgrade discussions should include a discussion about all parameters so the facility can plan appropriately. For example, a facility may be able to comply with a temperature TMDL through a facility upgrade only to find that a dissolved oxygen TMDL that limits nutrient loading will require them to remove their discharge during the critical period.

2.4.3.3 Bacteria TMDL Study and Analysis

A major challenge with bacteria is that levels are highly variable in the environment. Because of this, high levels of replicates are often collected in field studies - at least 20% and sometimes 50%. In one study 100% replicates were collected (two samples at every station for every survey).

The high variability of bacteria increases the uncertainty of the analysis. Due to the potentially high concentrations and volumes of bacteria that can be discharged from point sources or some uncontrolled nonpoint sources, they can cause receiving water bacteria levels to jump by orders of magnitude. For example, diffuse animal waste from a pasture may produces levels in the thousands whereas uncontrolled CSOs (combined sewer overflows, i.e., raw sewage mixed with stormwater flows) can produce bacteria levels in the hundreds of thousands combined with high flows. Therefore, problems with some large sources of bacteria tend to be identifiable despite the variability around individual samples.

Bacteria field studies are generally straight-forward. Samples are collected from the water body and potential sources. Flow may also be measured in the water body and in tributary sources to assist with calculating loads and mass balance. TMDL studies often include the sampling of storm events, especially the "first flush" of a rising hydrograph after a dry spell. The logistics of catching a storm event can be challenging. Other challenges include sampling from tidally affected areas, pump stations, CSOs, and other intermittent or dynamic sources.

Bacteria TMDL technical studies are usually based on a relatively simple analysis. The variability of data is somewhat offset by fact that the water quality standards utilize two criteria for bacteria – a geometric mean and 90% percentile. These criteria are typically reexpressed as statistical targets or percent reductions. And in many cases bacteria can be assumed to be conservative (no die-off or regrowth) for the purposes of calculating mass balances. Even when simple modeling is done that evaluates bacterial loading by land use, a bacteria TMDL can typically be completed using statistical tools in a spreadsheet program.

Two methods have been used for comparing bacteria to standards: (1) a non-parametric approach with original data; and (2) log-transformation of data and determining compliance from the fit to a log-normal distribution. Both methods have been used historically. The technical differences and appropriate situations for using each method can vary and are not discussed here

Most commonly the bacteria analysis is a combination of a mass balance analysis and use of the Statistical Roll-back (SRB) method. A mass balance analysis can help to identify nonpoint or other unidentified sources. The SRB method allows a simple percent reduction to be applied to both the geometric mean and 90th percentile values until both are meeting the criteria. This results in instream targets that are equal to the criterion for one metric and less than the criterion for the other metric. Commonly, the percent reductions are applied to nonpoint sources, while technology-based limits are set for point sources.

For modeling of bacteria in a tidal estuary, a dynamic two-dimensional model like WASP or GEMSS may be used. Due to the complexity of tidal actions and the human structures in tidal areas (levees and tide gates for example), determining nonpoint sources in tidal conditions can be difficult. In addition, tidal wetlands and sheltered areas attracting wildfowl and marine mammals may produce natural background levels of bacteria that are difficult to quantify. High natural bacteria levels can add to the stringency of point source limitations, already restricted by technology-based limits and shellfish protection requirement.

One complication in some bacteria TMDLs is when downstream criteria are more stringent than upstream. This occurs most often for a river flowing into marine waters, since marine standards are much tighter than freshwater. In this situation the analysis falls into two categories:

- Marine standards usually apply when salinity reaches the threshold value of 10 parts per thousand (ppt). This may require freshwater at the marine interface to have less bacteria than allowed by the freshwater criteria. To calculate a protective value for the freshwater boundary, a dilution ratio can be calculated from the background salinity of the marine waters, and the target calculated from that ratio. For example, if the Sound is 20 ppt, then the freshwater will be at 2:1 dilution when it has to meet marine standards, and therefore the the TMDL may base Loading Capacity for freshwater at a level double the marine standards.
- In some cases there may be less stringent criteria near the mouth of a river than upstream (for example, a river flowing into an estuarine industrial area such as the Puyallup River), or it may be possible to demonstrate from monitoring data that the river dilutes quickly to background as it enters the estuary. In this case, the TMDL can include an analysis of data to document that meeting freshwater standards in the river is still protective of marine standards.

In some situations there may be evidence that bacteria are not conservative, in other words they may grow or decay. Here are a few situations that have been observed in the field or are cited in literature:

- Bacteria may have a natural die-off rate. This is often applied as a first-order decay in modeling. The die-off rate may be site-specific, and it is usually much higher in marine water than in fresh water.
- Bacteria die-off may be enhanced by solar radiation.
- Bacteria adsorption to solids may reduce the die-off rate and subject bacteria to settling with the sediment. This may be of particular importance where settling may occur over shellfish beds or where resuspension of contaminated sediments may occur from wind or currents.
- Bacteria may regrow in sediments. This may particularly occur in sediments rich in organics and nutrients, for example in wetlands.
- Bacteria may grow in nutrient-rich effluent. There is some evidence of this occurring in fish-processing wastewater and municipal storm sewer systems experiencing nutrient-rich inputs.

In general, unless site-specific data suggest otherwise, the assumption that bacteria is a conservative parameter is also a conservative assumption. In most cases some die-off occurs, so not including die-off will result in protective TMDL allocations. But if data show regrowth or resuspension, then the margin of safety may not exist and those conditions should be taken into account in the TMDL analysis.

Natural conditions for bacteria can sometimes be an additional complexity. Background bacteria can range from nonexistent (Skagit River low flows, for example) or can be quite high where wildlife or wild fowl are present. Situations in past studies have included bacteria from elk herds, migratory bird concentrations, and flushing from wetlands during rain events. Also some kinds of bacteria sources, such as pigeons in culverts and bridges, present difficult questions regarding what is "natural". Separating natural background from human bacteria sources is challenging, and generally has to be handled as a site-specific issue based on available data and on-the-ground conditions.

Bacteria TMDLs in most cases will have seasonal or flow-based limits. Sources and transport processes can be quite different for dry season/low flow conditions and for wet season/stormwater wash-off conditions. At low flow, bacteria levels are likely to be dominated by direct animal access, poor manure handling (manure gun overspray, for example), and point source process discharges. At high flows, flushing of surface deposition into stormwater flows is the dominant mechanism. Therefore critical flow conditions for bacteria may be both summer low flow and a first-flush storm event, depending on the season and contamination pathway.

For permittees, low flow conditions will likely be more challenging for a steady-state discharge of treated wastewater, while high flow conditions may be difficult for stormwater and combined sewer overflows. The load-duration curve method has been used successfully in some TMDLs to develop flow-based effluent bacteria levels.

Typically several issues are taken under consideration for NPDES permittees in determining effluent loading for use in bacteria TMDL Loading Capacity scenarios:

- If a permit already contains bacteria effluent limits, are the limits sufficient to protect water quality?
- If there are no limits, or if effluent limits are not sufficiently stringent, are actual performance levels sufficient to meet standards?
- If existing effluent limits or performance standards are insufficient to protect water quality standards, is there a well-established technology that can meet more stringent limitations (such as UV disinfection instead of chlorination)?
- Ultimately, bacteria effluent loading may be set to meet the bacteria criteria at the end of pipe in order to meet standards. This may be applied when data on bacteria from a source are unavailable, or if a source is expected to have little or no bacteria in its effluent.
- If a facility has no normal source of bacteria, or if an existing permit has a no discharge provision (such as for the Dairy General Permit or other permit for land application), then effluent loading would be zero, i.e., no discharge, or a discharge with a concentration of zero.

For municipal stormwater permits, a watershed model may be used to develop loading estimates and TMDL targets. Often due to a limited amount of data, a simple model (such as the "Simple Method") may be appropriate. In cases where the TMDL covers contiguous permitted and non-permitted areas, loading levels may be set using one limit that applies as a WLA in permitted areas and as a LA in unpermitted areas, set perhaps in terms of load reductions or load per acre.

Permit managers should work closely with the TMDL technical and regional leads if an allocation less than criteria is contemplated. There may be flexibility regarding the values being set in allocations (concentration or percent reduction targets) and the values being included in the implementation plan as clean-up targets.

2.4.3.4 Temperature TMDL Study and Analysis

Developing a TMDL for temperature presents unique challenges for a number of reasons:

- The "pollutant" is primarily solar radiation, which is "natural." Therefore the impacts are caused by human activities that enhance the effect of solar radiation, such as removing shading vegetation, reducing flow, or modifying channel geometry.
- All water has a thermal load. Therefore all water sources need to be addressed.
- Although the physics of thermal capacity, transport, and exchange are fairly well understood and simple to model, temperature in a flowing waterbody is highly dynamic

A detailed discussion of temperature processes can be found here: Temperature - Overview of Stream Heating Processes. This is often included in an appendix to a TMDL study report, and could also be included in a permit fact sheet. Although the core of a temperature TMDL is the shade analysis, point source WLAs are very often developed using mixing zone analyses (discussed in more detail below). Shade information can be incorporated into a temperature model or used directly as "shade curves." The shade analysis is then combined with information about heat loads from point sources, flow, and channel morphology.

A temperature field study generally consists of:

- Monitoring over the summer with continuous temperature sensors. The sensors now available are small and inexpensive, so their placement in surface waters, as well as in outfalls or within a facility process train, is mainly limited by access and other field logistics.
- A field shade assessment, usually using hemiview digital photography, LiDAR, orthophotography or other methods of physical shade measurement.
- Flow measurements and evaluation of stream geomorphology (such as stream depth and width, disturbed area width, riparian canopy height and width, and streambed composition).
- Typically August is targeted to capture the hottest air temperatures and late June/July to capture the most intense solar radiation. However, in some situations a "shoulder season" is also targeted, such as September, when supplemental spawning criteria take effect in some waterbodies. In some cases, monitoring may be conducted from spring through fall or for an entire year.

As with all field data collection studies, data need to be analyzed for meeting QA targets. Then the analysis will follow several steps:

- The effective shade along the water body is calculated using a variety of tools including "SolRad", "Sunrise/Sunset", "Shade", and "tTools".
- Point source heat loads are calculated from measured flows and temperatures or are estimated from similar facilities.

- The shade values are then usually input into a stream temperature model. There are a variety of models used for temperature modeling:
    - QUAL2Kw is the most commonly used modeling framework. It provides a steady-state one-dimensional model with a diel heat budget.
    - rTemp is a simple one-dimensional heat response model that can calculate long-term temperature time series for small streams.
    - CE-QUAL-W2 may be used for complex situations, such as reservoirs or lakes, where a dynamic two-dimensional model (vertical and longitudinal) is needed.
    - Other models are available for use in studies, such as SNTEMP, SSTEMP, HSPF, GEMSS, and WASP.
- Some temperature TMDLs use "shade curves" to develop shade allocations. In these cases there may not be a formal model, and point source inputs may be addressed with end-of-pipe or mixing zone analyses. When shade and QUAL2KW modeling are performed,loadings from point sources are included. However, it is very common for mixing zone analysis to drive development of WLAs. The permit manager should work closely with the TMDL technical and regional leads to ensure that the permitted sources can address the TMDL appropriately, typically in the next permit cycle.

A critical element of most temperature TMDLs is the need to determine system potential (as an estimate of natural conditions). A water body listed as impaired for temperature in the 303d list (Category 5) may have exceeded the values set in state criteria before any human influences in the watershed. Washington's WQ standards account for this by limiting human-caused temperature increases to 0.3 ^{o}C when natural conditions exceed criteria. In addition, when a water body does not meet its assigned criteria due to natural climatic or landscape attributes, the natural conditions constitute the water quality criteria [WAC 173-201a-260(1)(a)]. We currently refer to our estimate of natural temperature conditions the "system potential."

There are different requirements for incremental increases when natural conditions are determined to fall below the temperature criterion. The incremental warming limits in WAC 173-201A-200(1)(c)(ii) and WAC 173-201A-210(1)(c)(ii) are intended to be applied directly to point source controls and not to the waterbody covered by the TMDL (Ecology, 2010). However, there are a few specific cases where incremental increases are provided under special conditions for the water body (for example, for the Pend Oreille River). In this situation, the TMDL may include additional analysis to determine when to apply incremental increases because natural conditions are are estimated to be below criteria.

TMDLs use modeling to estimate natural conditions by returning site-potential riparian shade and removing all human thermal loading. However, these estimates have usually not addressed channel morphology, changes in flow or groundwater inputs, or loss of large woody debris in great detail. True *natural conditions* (absent all human impacts) are impossible to fully assess. TMDL analyses can only estimate natural conditions, and the quality of that estimate is dependent on available data and resources. In a few cases historical temperatures are available to provide that estimate.

The determination of natural conditions has been thrown into confusion by a court decision in Oregon that rejected EPA's approval of natural conditions criterion in the Oregon WQ standards based on the judges determination of how Oregon applied natural conditions. The judge in Oregon's process found:

- Their natural conditions determination supplants the numeric criterion with a new criterion not established by rule.
- Their natural conditions determination is based on limited and uncertain data. Analyses usually ignore or discount a variety of watershed alterations that affect temperature, such as channel morphology, water clarity, or microclimate.

In Washington our natural condition determination does not supplant the numeric criterion and our modeling often involves greater complexity of data inputs so it may not run into these same concerns. However, this issue is still under review by water quality program. Note that for several other parameters such as dissolved oxygen, arsenic, etc…. Oregon's "general natural conditions criterion" was also rejected by EPA.

Temperature TMDLs should address two or three critical periods: (1) late summer conditions of critically high maximum air temperatures and critical low flow; (2) median summer air temperature and low flow conditions; and (3) early summer conditions of high air temperatures and high solar radiation levels. Multiple critical periods need to be evaluated because under median conditions (which may be much cooler than a hot, dry summer) the amount of incremental heating from human causes may be greater. In other words, a hot, dry summer will still be extreme in a natural situation but the past and present activities of people aren't going to change it that much. In a median situation the potential to increase temperature from a relatively cool natural baseline is much greater.

Detailed guidance for implementing a temperature TMDL is described in the document *Water Quality Program Guidance Manual – Implementing the State's Temperature Standards through NPDES Permits* (Ecology, 2010). This document addresses issues such as:

- Evaluating a discharge
- Determining "reasonable potential"
- Extended mixing zones
- Options to reduce thermal impacts from municipal wastewater treatment plants
- Monitoring considerations

How an NPDES-permitted point source will be affected by a temperature TMDL depends on the relative temperature of the source and the receiving water and the amount of dilution. Permitted sources that often are warm enough to increase the heat of the receiving water include cooling or wash water discharges and sources using uncovered tanks or lagoons for treatment. However, conventional treatment plants may still be warmer than relatively cold receiving water. Stormwater may also be warmer than receiving water if it is heated from running over pavement or from sitting in an exposed stormwater pond. The impact of any discharge will be greater on receiving waters with small critical dilution factors, such as a small stream, lake, or poorly flushed estuary.

Effluent thermal loading values for Loading Capacity scenarios may be based on a variety of approaches, including:

- Effluent thermal loading may be set on performance, measured or calculated.
- A cumulative increase from permitted outfalls of no more than 0.3 degrees Celsius at the edge of a mixing zone, assuming upstream temperatures at criteria. Note that the compliance time for implementation of point source WLAs may be shorter than the compliance time for the shade restoration needed to allow upstream conditions to meet criteria.
- A cumulative increase from permitted outfalls of no more than 0.3 ^{o}C with no mixing zone allowance (or in some cases 0.2 ^{o}C, allowing a margin of safety), assuming upstream temperatures at criteria.
- Permitted discharges are provided limitations equivalent to water quality criteria at end-of-pipe.
- If upstream temperatures are below criteria, effluent temperatures calculated from the equation provided in the standards (for example "28/(T + 7)") should be evaluated and applied if they are the most protective approach.
- The approaches described above could be applied using seasonal or flow-based limits.
- Effluent thermal loading may be set to zero, or in other words, no discharge during critical conditions.

Other options have been used in the past, including setting upstream conditions to observed or natural conditions. There are problems with these approaches and should be avoided, or only used after consultation between the technical and regional leads, permit manager, and WQP policy leads.

Permit managers can help ensure that several important issues are addressed in setting loading for permitted point sources in a temperature TMDL:

- All permitted sources (and possibly future sources) need to be included.
- Background water quality conditions should be clearly defined.
- Dilution factors should be harmonized between the TMDL and NPDES permit.

See the section below about WLAs for more discussion of how to set loading limits in a TMDL.

2.4.3.5 TMDL Study and Analysis for Dissolved Oxygen, pH, or Nutrients

Changes in DO, pH, and nutrient levels in surface waters are usually interrelated and are often combined in one TMDL study and report. Relatively complex studies evaluate these interacting factors. These can include:

- The oxidation of organic carbon (CBOD), organic nitrogen, or ammonia.
- Water temperature effects on dissolved oxygen saturation.

- Primary productivity, its stimulation by human nutrient discharges, and its effect on DO and pH levels in the water body. Primary productivity can result from phytoplankton (suspended algae), periphyton (attached algae), or macrophytes (emergent aquatic vegetation).
- Algae and vegetation species composition and growth cycles.
- Food chain effects, such as zooplankton predation.
- Sediment interactions, such as settling of phytoplankton and the mineralization and rerelease of nutrients ("internal loading").
- Aesthetic values, such as water clarity, color, nuisance vegetation, or exotic species.
- Toxicity of ammonia or toxic algae blooms.

This kind of TMDL is typically triggered by a DO or pH 303(d) listing, but there are also some 303(d) nutrient listings. Each water body has some unique combination of the issues listed above. In some cases it may be a simple BOD TMDL. In other cases nutrients are a contributing or primary factor. DO may be below criteria, but pH may be acceptable. Additional factors may be uncovered as the study progresses.

Field studies for DO/pH/nutrient TMDLs are generally logistically challenging and time intensive. A broad number or parameters need to be measured. This includes samples collected for laboratory analysis, field measurements (grab and continuous), time of travel, and flow. Additional special studies may be needed, such as productivity, algal species identification, reaeration rates, light attenuation, ground water quality and exchange rates, or sediment oxygen demand. Surveys (often referred to as synoptic surveys) may require multiple teams and may need to be repeated on multiple dates. Typically these surveys involve intensive diel data collection usually performed two to three times during the critical season (summer), although some studies may collect regular monthly or bi-weekly samples.

For this kind of TMDL, support from the permit manager is valuable. Sampling from NPDES dischargers may be necessary and can be complex. Some permittees may be already collecting necessary data, but they may need to provide the original data and not averaged values. Additional monitoring may still be necessary to improve the quality of the model. Early coordination between permit managers and the TMDL technical and regional leads is important for the success of the study.

Field study structure and methods will depend heavily on the model framework selected.

- For a one-dimensional stream model with diel variation, QUAL2KW is most commonly used.
- For two-dimensional modeling (vertical and longitudinal):
  - WASP can be used in a river or estuary with steady-state flow, tidal boundary, or linked to a hydrodynamic model.
  - CE-QUAL-W2 can be used as a hydrodynamic model where a lake or reservoir is present.
- For three-dimensional dynamic modeling, GEMSS has most often been used.

Based on the model chosen, field monitoring will need to be structured to collect data at the appropriate spatial and temporal resolution and timing to meet model input needs. A steady-state model can work with a relatively simple plan, while a dynamic model may require monitoring that follows flow or tides, and a multi-dimensional model may require samples at multiple depths and horizontal locations.

The models for these TMDLs can be the most challenging to calibrate. Typically they follow this process:

1. Flow and hydrodynamics: For steady-state models a pre-calculated flow data set is entered. For dynamic models flow may need to be calibrated to observed data at gaging stations. For an estuarine model, the model also needs to be calibrated to the downstream tidal conditions.
2. Temperature: Again, for a steady-state model the temperature time series may be input, or the model may have a dynamic temperature model which must be calibrated.
3. Full water quality model: Depending on the complexity of the model, calibration may be based simply on BOD and oxygen levels measured in the waterbody, or it may also depend on nutrient and chlorophyll-a levels.

With so many dynamic factors, calibration is a complex trial-and-error process with many iterations. Modelers must use their judgment to find a balanced calibration of multiple parameters at several locations and possibly over a period of time. Generally the modeler's goal will be to calibrate the parameter being analyzed for the TMDL as closely as possible, while allowing more variability for other parameters. Parameters such as chlorophyll-a, which are highly variable in the field and difficult to measure with accuracy, will be the ones least expected to calibrate closely.

The model calibration process often includes the estimation of sources that are poorly quantified or unidentified. There is usually more certainty around the flow and concentrations of pollutants from point source discharges, although this is not always true. Some point sources are highly variable, or monitoring data may be sparse or nonexistent. Uncertainty in nonpoint sources can be found for ground water inflows, small tributary inflows, and unknown sources. Part of the forensic aspect of modeling is to determine whether an unidentified source needs to be included, input values for poorly quantified sources should be changed, or whether potential sources are indistinguishable from the overall variability of the model.

DO/pH/nutrient TMDLs generally require a model run to estimate natural conditions. This scenario can verify when criteria can be met in the absence of human impacts, and when DO or pH levels are determined to exceed criteria naturally. Then the requirements for incremental change due to human activities will apply when natural conditions do not meet the allowed criteria.

Note that the complications from the Oregon court case described above for temperature natural conditions may apply here as well. If the TMDL includes an analysis of natural conditions, consultation between the technical and regional leads, permit manager, and WQP policy leads is advised.

Additional guidance for applying DO standards in a TMDL can be found in Guideline #1 of Ecology (1996).

Determining loading reductions for Loading Capacity scenarios can be challenging. Based on the calibration and natural conditions scenario, the proportion of loading reductions for nonpoint and point sources must be determined, which can be potentially controversial.

The permit manager can work with the TMDL technical lead to evaluate reductions in point source loading in step-wise fashion:

- Examine how the system responds when discharges are at design levels or maximum permit limitations. This can give some indication of the sensitivity of the system to the discharge.
- Evaluate the effect of loadings at full design flows with existing permit limits and consider future expansion plans
- Evaluate performance-based limits to see if a smaller loading than allowed by existing permit limits is possible.
- Explore whether reduced limits based on known available technology can be applied to different point sources.
- When estimated natural conditions exceed criteria and only small incremental change is allowed, potential discharge levels may be very low. As a result, treatment costs may be high, or a WLA may be set to zero to provide a margin of safety. The permit manager should work closely with the TMDL technical and regional lead to explain the potential effect on the permittee(s). Where removal of a discharge or significant facility modifications are a potential short- or long-term necessity to meet standards, the TMDL team should notify and work with their respective supervisors.

Guidance on effluent loading for DO modeling can be found in Guidelines #4 and #5 of Ecology (1996).

2.4.3.6 TMDL Study and Analysis for Sediments and Turbidity

Sediments and turbidity are predominantly a nonpoint source issue. Technology-based treatment limits are usually adequate to meet TMDL Loading Capacity. In a few cases industrial point sources may be a significant source of sediment or turbidity loading and require reductions in a TMDL.

Standard best management practices for stormwater permit limits can usually meet TMDL Loading Capacity. However, implementation and compliance work is often needed to ensure sediment and turbidity reductions in permitted municipal, industrial, and construction stormwater.

A TMDL study for turbidity or sediment is straight-forward. Turbidity can be measured by laboratory samples or field monitoring. Continuous monitoring for turbidity is now available. Sediment is measured by laboratory measurement of total suspended solids (TSS).

Several different approaches to the analysis have been used in past TMDLs:

- TSS and turbidity are tested for the strength of their regression relationships.
- For a TMDL/toxics TMDL (Upper and Lower Yakima River TMDLs, for example), the relationships between the TSS and the toxic of concern are evaluated.
- Relationships between TSS, turbidity, and flow are evaluated.
- TSS can be compared to biological indices to see impairment thresholds (Little Spokane River and Hangman Creek TMDLs).
- Using mass balance and statistical analyses, sources are quantified, and then sediment reductions calculated to meet turbidity or toxic criteria or biological thresholds.

One particular challenge with sediment and turbidity TMDLs is the determination of natural background levels. Natural sediment sources are likely to be diffuse, dynamic, and highly variable. Therefore, identifying and quantifying background sediment levels for comparison to human-caused increases in sediment above background can be challenging and potentially controversial.

If a point source is to significantly contributing to sediment or turbidity impairment, reductions may be calculated for that source. For the four TMDLs done historically, only one source was found to have permit limitations requiring reductions, and that source was inactive at the time of the TMDL.

2.4.3.7 TMDL Study and Analysis for Toxics

Toxic parameters in many ways present the most complex challenges for TMDL studies. A number of factors contribute to this complexity:

- The impairments on the 303(d)-list for toxic compounds are based on acute and chronic water quality criteria from both the State WQ standards and EPA National Toxic Rule. However, only a few toxic parameters have criteria, and water quality impairments may be linked to other toxics not listed (for example, PBDEs).
- Toxics effects may occur at very low concentrations. High levels of field method quality and extremely sensitive laboratory methods are required for these parameters. Some criteria may be so low that it becomes very difficult to measure that parameter in the environment, and non-detection values may not necessarily mean compliance with standards.
- Some criteria may depend on ancillary parameters, such as metals whose toxicities vary with hardness.
- Some toxics have diffuse and hard to control sources (for example, PCBs from atmospheric deposition or consumer products) which require an aggressive nonpoint control strategy. This can put pressure on permitted dischargers if they are viewed as “easy to regulate”, as compared to the low level of “reasonable assurance” for nonpoint source reductions.
- It is very expensive to characterize seasonality and nonpoint sources for toxic chemicals, which produces uncertainty in determining loading capacity.

- Toxic compounds can be lipophilic (attach to oils and fats) or hydrophilic (dissolve easily in water), which result in very different fate and transport mechanisms.
- Toxic compounds often bind to sediments and are addressed by separate sediment quality standards.
- Toxic compounds can bioaccumulate and bioconcentrate, moving up the food chain at higher and higher tissue concentrations in the biota. Specific human health criteria have been developed for some of these compounds.
- Achieving compliance with the standards within the 10-year compliance schedule allowed by the State WQ standards can be very difficult. It often takes multiple decades to achieve reductions for persistent legacy pollutants like DDT and PCBs.
- New information on toxicity is constantly emerging from scientific research. However, toxicity thresholds from research may not yet be adopted as criteria in state regulation. The water quality standards allow for new science to be applied through the narrative criteria protecting human health and aquatic life. This approach is not commonly used, since it depends on documentation of the research and making the case that literature values are applicable to a specific situation.

Toxics are found in the environment in a variety of forms:

- Toxics dissolved in the water column.
- Toxics with sediment fractions.
- Bio-accumulative toxins.

Depending on the parameter, TMDLs address these different forms in different ways.

In all cases, the involvement of the permit manager is critical. Determining toxic loading from permitted sources may be difficult and expensive, but important to a successful study. Evaluating ways to reduce the discharge of toxics from permitted sources may be complex and controversial. At both the study phase and the TMDL development phase, the participation of the permit manager can help to avoid pitfalls and ensure a TMDL that is both protective and feasible to implement.

2.4.3.7.1 TMDLs for Dissolved Toxics

The simplest toxics TMDLs have simple concentration-based criteria and a simple reactions and transport mechanisms. Parameters addressed with this approach include ammonia, chlorine, and some metals. Field studies generally will measure receiving water and effluent parameter levels, flow, pH (for ammonia), hardness (for metals), and temperature.

Only a few TMDLs from the 1990s were done for chlorine and ammonia, and these were TMDLs for a single discharger. TMDLs were based on dilution at the edge of the mixing zone and on background pH for ammonia criteria. TMDLs for these parameters using this methodology will likely be rare in the future.

In general, other parameters being addressed with water quality standards for dissolved contaminants are approached the same way. Effluent levels are mixed with upstream receiving water at the edge of a regulatory mixing zone. Effluent limits for Loading Capacity scenarios are calculated from levels that allow criteria to be met at the edge of the mixing zone.

One challenge with many of the metals (such as cadmium, lead, and zinc) is that criteria are based on hardness. Therefore criteria need to be assessed in the context of the hardness of the mixture of effluent and upstream flow.

Also, many of the criteria for dissolved metals are based on dissolved fraction, and may need to be translated into total recoverable concentrations for TMDL allocations and for use in the permit. Conversion factors are available in federal criteria, or site-specific conversion factors can be calculated from sampling results.

Guidance on applying metals standards in TMDLs is in Guideline #6 of Ecology (1996).

There are other complex and controversial issues regarding the toxics criteria in regulation and the laboratory method used to determine compliance with criteria. These issues create difficulties both for receiving water sampling and effluent sampling. For example, periodically there have been attempts by permittees to determine “bio-available” toxics. However, there are concerns that using bio-available toxics to set permit limits results in higher effluent limits that may not provide a margin of safety. The TMDL process calls for an approach that is protective of the environment and human health, and usually TMDLs are based on the total recoverable amounts of the compound in question.

In addition, there are often more modern analytical methods for toxics that have not been adopted into regulation. For example the only method allowed by regulations to assess compliance with a PCB limit is EPA Method 608, which has a relatively high detection limit. One approach to address this problem was applied in the City of Spokane NPDES permit, where EPA Method 1668 (HR/GC-MS) was required for monitoring to establish a performance-based PCB effluent limitation and a Toxics Management Plan.

For point sources there will likely be a lot of interest in background levels. Although some upstream sources may be subject to pollutant allocations, in other cases they are beyond Ecology’s regulatory control. In these cases, a high background level may force more stringent limits for point sources.

For example, metals in the Spokane River come largely from mining in Idaho, and the background levels arriving from out-of-state have an impact on allowable discharges from Washington sources. Geologic sources from erosion or farfield atmospheric deposition may also be issues. In some cases sources in British Columbia may be affecting Washington water quality. All of these factors add complexity to the TMDL analysis. And if natural levels are sufficiently high (for example, arsenic in the Stillaguamish River), the TMDL may not allow additional discharge of loading of that contaminant, and point sources may have to comply with zero wasteload allocations (no discharge allowed).
Some metals TMDLs are based on the dissolved fraction. However, metals, and many other

compounds, are likely to adsorb onto sediments. Therefore a TMDL may need to address both dissolved and sediment fractions to capture transport and fate processes.

2.4.3.7.2 TMDLs for Toxics with Sediment Fractions

TMDLs for toxics may focus on both water column and sediment. Monitoring for the TMDL will generally look at dissolved and particulate fractions in discharges, in the water column, and where appropriate, sediment contaminant levels in depositional areas. Flow, sediment fractions, organic carbon, temperature, and pH will affect transport processes and toxicity. Priority pollutant scans and outfall sediment studies required on a periodic basis for municipal WWTPs can provide important data to help support sediment TMDLs.

Some TMDL studies of these toxic chemicals will work backwards from the two pathways. Beginning with sediment quality standards, the study evaluates sediment levels, sediment adsorption rates and settling rates, water column levels, and discharge levels to determine loading limits for toxic compounds that are protective of sediment standards. For Loading Capacity scenarios, those levels are compared to the loading reductions needed to meet water quality criteria, and the more restrictive loading limits are selected. This approach has been applied in Bellingham Bay.

TMDL studies for toxics, especially for DDT metabolites and other pesticides, will often look at the statistical relationship between sediment (as measured by TSS or turbidity) and the contaminant. For toxics that strongly adsorb to sediments, the sediments themselves may be the target of pollutant allocations based on associated levels of the contaminant and its criteria. This has a practical aspect, since sediments may be simpler to manage than the contaminants themselves. The Lower Yakima River was an early example where this approach has been used successfully.

Typically TMDLs using this approach of monitoring and managing sediment toxics levels have focused on agricultural runoff. Point sources have been determined to be minor sources, and Loading Capacity scenarios use technology-based or performance-based sediment effluent limits.

One complication of this method is that the sediment-contaminant relationship may change over time. Therefore long-term monitoring and adjustment of targets may be needed. Also for stormwater, the effectiveness of BMPs for controlling both sediment and contaminants needs to be monitored and assessed.

Another problem is that this approach is not workable in areas with low suspended solids. The Okanogan and Spokane Rivers are examples of water bodies with listings for lipophilic compounds like PCBs but very low solids. Thus, the approach used in a toxics TMDL needs to be customized based on the watershed characteristics.

2.4.3.7.3 TMDLs for Bioaccumulative Toxics

Many organic compounds, like DDT and PCBs, and some metals, like mercury and arsenic, are bioaccumulative. Therefore, in addition to their direct toxicity in the water column and their potential to adsorb to sediments, they also can concentrate as they pass up the food chain. This adds an additional level of complexity to these TMDL studies.

Elevated fish tissue concentrations of toxic chemicals can trigger a 303(d) (Category 5) listing. To set point source effluent limits for Loading Capacity scenarios in a TMDL for bioaccumulative toxins, the methods for dissolved and sediment fractions are combined with a food chain model. Human health criteria for TMDL compounds are applied to fish tissue, and then levels in the receiving water can be back-calculated with literature bioaccumulation rates and a food chain model. Using the model, water column concentration targets can be determined that will protect human health criteria in fish tissue, and then effluent limits can be calculated that meet the water column concentration targets.

The fish consumption rate used for linking human health criteria to contaminant levels in the environment and in pollutant sources is currently under review. As can be seen from this discussion, a higher fish consumption rate can lead to more stringent permit effluent limits. This is a highly controversial issue and outside the TMDL process, which relies on the regulations in place at the time of TMDL development and adoption. However, the permit manager may want to monitor this issue because of its potential impacts on permit management.

Field studies for bioaccumulation generally require additional sampling of tissue from fish or other biota. Another tool used in more recent studies for assessing low-level concentration of lipophilic toxins is the semipermeable membrane device (SPMD). The SPMD contains a lipid material that collects the contaminants. The methodology for this device allows correction for site conditions and back-calculation to water column concentrations.

In general, challenges for point sources for toxic TMDLs include:

- The complexity of the methodology.
- Criteria and targets that may be much lower than laboratory detection levels.
- The cost of laboratory analysis and uncertainty around appropriate methods.
- The dependence of effluent limits on background levels.
- The uncertainty of controlling nonpoint sources of toxics.
- The uncertainty of BMP effectiveness to reduce toxics in stormwater.
- The difficulty in meeting stringent effluent limits with treatment.

2.5 Establishing Wasteload and Load Allocations

2.5.1 General Approach

The amount of time and effort necessary to determine TMDL allocations will depend on a variety of factors:

- The complexity of the TMDL study.
- The amount of reductions needed to meet standards.
- The number of identified or unidentified sources, and their relative contributions to the impairments.

- The availability of “low-hanging fruit”, i.e., pollution reductions solutions that are relatively well-established, affordable, and noncontroversial.
- The amount of controversy and political sensitivity that is aroused by potential implementation activities.

No specific approach can be described for how allocations are developed. Because Ecology permit writers do water quality-based analyses as part of basic permit writing, TMDL analyses may mirror ones already done during permit development. However, each TMDL is unique with regard to its parameters, environmental dynamics, dischargers, and land uses and TMDL studies frequently add new information to previous water quality based analyses.

Answers to several general questions can help provide direction for preparing allocations:

- What are the proportions of nonpoint versus point source? If point source contributions are very small, then existing permit limits may be sufficient. Performance-based limits should also be evaluated if they reflect the true situation and can provide reduced loading at no cost. For example, municipal WWTPs with UV disinfection can meet far lower limits than conventional technology-based permit limits.
- Are there clear opportunities for applying standard and accepted BMPs for nonpoint source reductions based on observed conditions? For example, there may be documented opportunities to improve livestock or stormwater management where no permit is in effect.
- Are there well-known and advanced treatment methods, process changes, or water reclamation opportunities that are affordable and could allow lower loading limits for significant point sources?

Table E.1 provides an overview of WLA options, along with intent and history, examples, and the benefits and drawbacks of approaches. Every NPDES permit in the TMDL footprint should receive a WLA based on one of these choices in Table E.1. It is important that the permit manager work with the TMDL technical and regional leads to ensure that the WLA is consistent with meeting the requirements of the TMDL, the technical realities of the permitted discharger, and the practicality of including the WLAs in permit limitations. Recommended approaches are provided for each category but may not represent all options available to permit writers and TMDL developers.

Table E.1 Summary of Wasteload Allocation Approaches

<table>
<tr><th></th><th>Approach</th><th>Intent and history of approach</th><th>Example</th><th>Benefits and challenges with this approach</th></tr>
<tr><td rowspan="2">1.</td><td>No WLA provided</td><td>These NPDES sources are considered insignificant or nonexistent. The NPDES dischargers will not be regulated by this TMDL, effectively “outside” of the TMDL. The NPDES permit will either rely on existing language or be silent for this parameter.
Historically, some TMDLs have used this approach. However court cases (such as the “Pinto Creek” decision) are suggesting this is no longer legally acceptable.
A related problem is a new discharge that did not exist when the TMDL was developed. If no reserve for future growth is set aside, it is unclear how a new discharger proceeds.</td><td>A non-contact cooling water discharge in a Bacteria TMDL</td><td>This is attractive as a “low maintenance” approach (Ecology basically just ignores these sources in terms of TMDL implementation). However, under recent court rulings it could be interpreted as a WLA of zero.
Also, this leaves a source outside the TMDL, when it may be important to regulate the source. It can also create liability for a discharger, if they are not included in the TMDL, because they might be open to legal action for discharge to an impaired water body without being covered by a WLA. This is especially true for temperature.</td></tr>
<tr><td colspan="4">1. This approach is no longer acceptable. All TMDLs in the future should identify existing NPDES permits and include them in wasteload allocations (WLAs). Should a TMDL recommend a WLA of zero that should be explicitly stated in the TMDL (see option 6 below for this approach)).
2. All TMDLs should explain how a new NPDES discharge gets included in the TMDL.</td></tr>
</table>

Table E.1 Continued

<table>
<tr><th></th><th>Approach</th><th>Intent and history of approach</th><th>Example</th><th>Benefits and challenges with this approach</th></tr>
<tr><td>2.</td><td>Technology-based WLA</td><td>WLA is based on existing technology-based permit limits. This would usually apply when existing permit limitations are adequate to be protective of water quality standards.</td><td>Small WWTP in a large river</td><td>NPDES discharges get legal clarity and are subject to the TMDL, ensuring that water quality standards are met, and permittees don’t have to change their current permit limitations or operations. However, this may provide no room for future growth past technology-based limits.</td></tr>
<tr><td></td><td colspan="4">Recommended Approach:
1. This approach should be applied when a discharge has an existing permit limit or is covered by a general permit which is sufficient to not contribute to an impairment addressed by a TMDL.
2. The TMDL needs to specify a numeric WLA that covers each discharge.
3. If existing permit limits are using a significant fraction of the capacity in a TMDL, the discharge should be reviewed for possibly tightening the limits to performance-based, which could provide capacity for other sources or future growth.</td></tr>
<tr><td rowspan="2">3.</td><td>Performance-based WLA</td><td>WLA is based on past performance. This may represent a reduction in past effluent limits, or it may be applied when no effluent limits are in place. This is the preferred approach for a discharge with an insignificant/nonexistent load and is applied in many TMDLs.</td><td>WWTP with UV disinfection</td><td>NPDES discharges get legal clarity and are subject to the TMDL, ensuring that water quality standards are met. Permit limits are tightened, but operations don’t need to change. This is useful as an interim step while additional load reductions are implemented. However, it allows no room for future growth and may not account for the full range of discharge variability.</td></tr>
<tr><td colspan="4">Recommended Approach:
1. Apply to discharges that are insignificant or whose existing performance is adequate to meet the TMDL.
2. Apply when tighter limits with existing treatment provide capacity or when additional treatment reduces pollutant loads below technology-based limits. In this case, new limits may be placed in an individual permit. For a general permit, the TMDL WLA may be specified in an appendix, or an individual permit.
3. Useful as an interim step during a compliance schedule.</td></tr>
</table>

Table E.1 Continued

	Approach	Intent and history of approach	Example	Benefits and challenges with this approach
4.	**WLA based on meeting criteria "end-of-pipe" or at the edge of a Mixing Zone**	The WLA is based on criteria with or without a mixing zone. This approach is sometimes used when there is limited information about the discharge or if the discharge is not considered a significant source.	Bacteria in stormwater	This is used when necessary to comply with TMDL and meet WQ standards. It allows a WLA to be set with limited data or for an insignificant source. This approach works well for bacteria or temperature, where the WLA is the same parameter as the criteria. This would not work for parameters such as BOD or nutrients (to protect dissolved oxygen) or a bioaccumulative toxin.
	Recommended Approach: 1. Use for an appropriate parameter in situations of limited data or for insignificant sources. This may apply to either an individual permit or general permit.			
5.	**Water quality-based WLA**	TMDL requires reduced effluent limitations beyond past performance. This has been applied in some TMDLs.	WWTP w/ tertiary treatment to reduce BOD or nutrients	This is used when necessary to comply with TMDL and meet WQ standards. The disadvantage is the treatment costs for the discharger. Matching needed reductions to treatment options may be difficult.
	Recommended Approach: 1. Use when necessary to meet water quality standards.			
6.	**WLA = Zero**	TMDL bans the discharge of this pollutant. This could require either that a discharger will never discharge that contaminant or that the discharge is removed from the receiving water during critical periods. This has been applied in a few TMDLs for very sensitive waters or when a discharge goes to land application or full containment.	Chehalis R BOD limits. Industrial stormwater toxic limits, a sand and gravel facility that currently infiltrates stormwater during the TMDL critical period	This is used when necessary to comply with TMDL and meet water quality (WQ) standards. NPDES discharge gets legal clarity and is covered by TMDL. Zero discharge adds margin of safety because it eliminates risk of spill or plant upset. Disadvantages are dischargers' costs for treatment, monitoring, or alternative disposal and their risk of a possible violation.
	Recommended Approach: 1. Apply when the severity of the impairment warrants the elimination of a discharge. 2. Can also be used for a permittee that has eliminated its discharge through land application or a zero-discharge containment. 3. Do not use the term *zero discharge* when what is really meant is an insignificant discharge for which a performance-based WLA should apply (Option 3).			

EPA requires that all TMDLs include a *daily load* for all WLAs, based on a court case from the eastern U.S. where the judge said in effect “a daily load is a daily load”. For a variety of reasons, EPA recognizes that it might continue to be appropriate and necessary to identify non-daily allocations in TMDL development dispite the need to also identify daily loads (EPA, 2007).

WLAs should be set in terms that make sense from an implementation point of view, but to meet this procedural requirement, a daily load must also be provided. For bacteria, a concentration limit makes most sense, but a load can be calculated and provided to meet this requirement. The load used in a TMDL WLA may be based on a specific flow such as current discharge or design capacity and may need to be adjusted to reflect the actual conditions in the waterbody and at the discharge facility at the time of permit issuance.

The permit manager has a critical role to help ensure that WLAs in the TMDL:

- Correctly identify dischargers.
- Apply concentrations based on feasible treatment technology.
- Apply reasonable design or performance-based flows.
- Apply mixing zones and dilution factors that are consistent with established or anticipated permit conditions.
- Specify the point of compliance.
- Can be translated into reasonable, clear, and enforceable permit limits.
- Have considered WLA targets that are practical to monitor.

Ultimately it will be a team effort that will lead to a reasonable set of allocations that will meet TMDL goals. This requires the leadership of the TMDL regional lead, technical support from the TMDL technical lead, and information and advice from nonpoint specialists and permit managers. Ideally, this team effort will engage the constructive energies of stakeholders to help them see the goal and purpose of the TMDL and help find an acceptable outcome through collaboration. That being said, for those TMDLs that are contentious, the TMDL team can expect to devote significant time into setting allocations.

2.5.2 Small or Insignificant NPDES-Permitted Discharges

One very important point with WLAs is that every permitted discharge in the TMDL footprint should receive a numerical allocation, even if that WLA is zero. If an existing discharge is not mentioned in a TMDL, it creates confusion about whether it is meant to be performance-based with insignificant loading or truly a zero allocation. Current policy interpretation is that if a discharge has no allocation, it effectively has a zero allocation. Where we know that there is a potential for even a small discharge this creates an unecessary risk for the discharger. Permit managers have an important role in helping to ensure all permitted discharges affected by a TMDL receive a numerical WLA.

2.5.3 Bubble and Aggregated Allocations

An approach that permit managers might consider in dealing with WLAs for small loading sources is the *bubble allocation*. This approach creates one WLA which then can be applied to multiple dischargers. The bubble allocation can also be used for large dischargers to encourage efficiency and cooperation. The Willapa River Dissolved Oxygen TMDL has applied a bubble WLA for two municipal dischargers and three fish processing industrial discharges.

The advantages of this approach are that a bubble WLA:

- Is less cumbersome than multiple small WLAs.
- Reduces compliance risks for dischargers that have very small discharges .
- Creates an incentive for multiple dischargers to work together toward a common goal and may result in improved cooperation and efficiency.
- Creates an incentive for a new discharger to reduce loading to fit into the bubble WLA.

A similar approach applied to dischargers under a stormwater general permit is an aggregated allocation. In this case, an aggregated WLA may be provided to multiple discharges under different permits, provided they are in a similar category (municipal, or similar industries). An aggregated WLA may be structured by watershed or receiving water, rather than permit by permit. They may also address the WLA in terms of load reduction by area or percent load reductions, rather than specific loads.

The advantage of an aggregated WLA is that it provides a simpler approach when the quality and quantity of existing water quality data won't support multiple, more detailed WLAs. However, EPA encourages disaggregatation of permittees when "circumstances allow to facilitate implementation" and if "permit writers… have more detailed information…to effectively identify reductions for specific sources" (EPA, 2014).

Disadvantages of a bubble or aggregated WLA include:

- Bookkeeping can be challenging. Discharges should be tracked cumulatively to ensure the WLA is not exceeded.
- The appropriate level of monitoring would need to be determined.
- Compliance would need to be defined. If the bubble or aggregated WLA were exceeded, the individual dischargers that were responsible would need to be identified and held accountable.

2.5.4 New NPDES Permits

Permit managers may want to ensure the TMDLs establishes sufficient reserve capacity for future permitted dischargers created by economic development or population growth. Inclusion of a reserve for future growth in a TMDL may come at the expense of the capacity available for WLAs to existing discharges unless a reasonable level of compliance with proposed WLAs and load allocations (LAs) can be anticipated, allowing the creation of a set-aside for future growth.

Many TMDLs were established before the issuance of stormwater NPDES permits. And over time new sources or new geographic areas come under coverage of stormwater permits. In these cases if the stormwater loading was already part of the LAs, a portion of the LA can be applied to the stormwater WLA. For TMDLs already approved, a "correction letter" can be sent to EPA explaining that the loading is unchanged, but some areas have been shifted from LA to WLA.

For new TMDLs, language can be included that anticipates that shift and allows it to occur administratively. One approach that can help address this is to define loading as mass per acre, allowing a simple adjustment geographically. EPA has suggested the following language for inclusion in TMDL reports (Ramrakha, 2014):

> *Most stormwater in urbanized areas is conveyed through drainage systems called municipal separate storm sewer systems (MS4s) and is covered under the NPDES municipal stormwater general permit. There may be some existing stormwater sources of bacteria in the watershed that are not currently under NPDES permit. The allocations for such sources are expressed in the TMDL as the "load allocation" contingent on the source remaining unpermitted. However, this part of the load allocation could at some future time be deemed a "wasteload allocation" if the stormwater discharge from the source is required to obtain NPDES permit coverage.*

2.5.5 Bacteria WLAs

The issues discussed above for setting bacteria loads in a TMDL Loading Capacity scenario will guide the final selection of WLAs. Permit managers will need to work with the TMDL leads to develop a final package of WLAs and LAs, which will depend on determining:

- How seasonal or flow-based allocations might be applied.
- The levels of natural background bacteria.
- Whether technology- or performance-based WLAs are reasonable and sufficient to protect the receiving water, or whether more stringent limits are needed and feasible, such as meeting criteria at the edge of the mixing zone or at end-of-pipe.
- Which point sources can meet a zero WLA.
- The amount of loading reduction needed for nonpoint sources and the final LAs set.
- Whether a reserve for future growth is feasible or desirable.
- How to address LAs and WLAs for bacteria in stormwater.
- How a Margin of Safety was addressed.

2.5.6 Temperature WLAs

All permitted discharges need to have a WLA defined, because all discharges have a temperature. The TMDL can acknowledge the presence of dischargers that are insignificantly small or cooler than the receiving water and give them WLAs that allow them to continue their current operations at expected levels. The permit should implement these WLAs through a performance-based limit. Placing these WLAs into permits also puts them on notice that the receiving water is impaired for temperature and provides consequences for an unusual discharge of heated water.

If a temperature WLA is expected to create difficulties for the compliance of a permittee, then seasonal or flow-based WLAs (and permit limits can be explored. Seasonal limits are simpler to administer but need to take into account the full range of potential variability in that season. Limits based on flow-based WLAsare more difficult to administer, but they have been used very effectively to implement WLAs in a way that is both protective and reasonable.

Cumulative WLAs can be considered if technically reasonable for permittees with multiple outfalls. This could apply to stormwater, industrial process, or other kinds of permits. Again, this approach may provide some flexibility for compliance and can be applied with limited data, but it may also be more challenging to administer.

To meet EPA requirements, in a temperature TMDL point sources should have effluent temperature loading expressed in kilocalories per day or BTUs/day, calculated from design or performance-based flow and a target temperature. But it may still make more practical sense to also have a temperature limit, either absolute or as an incremental increase above background.

In many temperature TMDLs, especially those for larger river systems, shade modeling does not drive WLA development for individual permits. The magnitude of solar heat inputs dwarfs the input by the individual permittee. In those cases, mixing zone modeling is used to develop WLAs and ensure there are no water quality problems in the vicinity of the outfall.

2.5.7 DO, pH, and Nutrient WLAs

The process for exploring possible WLAs in a DO/pH/nutrient TMDL is described above as part of the TMDL study. Several methods have emerged from past TMDLs that can help establish WLAs and permit limits that are feasible for a permittee to meet:

- Evaluate opportunities for seasonal or flow-based WLAs and limits that would provide dischargers with some flexibility with compliance while still protecting water quality standards.
- Investigate advanced treatment technologies that may be feasible for lower effluent loading.
- Consider reclaiming and reusing effluent to allow a seasonal zero allocation (example: Upper Chehalis DO TMDL).
- Consider a BOD WLA and limit that combines carbonaceous BOD with ammonia BOD to provide flexibility in treatment plant operation (example: Snoqualmie River Multiparameter TMDL).
- Explore the possibility of a bubble allocation based on coordination between dischargers (example: Willapa River TMDL).

2.6 Public Outreach and Publication of the TMDL

TMDLs are typically prepared jointly between the EAP technical lead and the regional water quality TMDL lead.   There are two templates for this approach:

- A WQIR/IP document that combines the Water Quality Improvement Report with a detailed Implementation Plan. This is the default approach unless special circumstances warrant the use of two separate reports described in the next bullet.
- A joint Water Quality Improvement Report (TMDL/WQIR) that contains the technical study and an implementation strategy. This approach is combined with the publishing of a separate detailed implementation plan.

In some situations, the TMDL technical study is published separately. This may occur if there is a delay between the technical study and the final WQIP, or if a contractor or local partner conducted the technical study. A separate technical study report will usually be included as "Volume 2" in the final TMDL Submittal Report, although in some cases it may be summarized and referenced.

The regional TMDL lead usually convenes an advisory group early in the TMDL process to offer an opportunity to review the QAPP, discuss TMDL goals, and look for partnerships in TMDL development. After the technical portion of the TMDL is drafted by the EAP technical lead, the regional TMDL lead reconvenes the advisory group. The regional lead goes over the technical findings of the TMDL study and explores implementation actions. It is common to include municipal permittees as part of the advisory group. Consistent and clear messaging to the permittee by the regional lead and the permit writer is important to avoide confusion during TMDL preparation and eventually when permits are renewed.

After internal review by managers, the final draft WQIP/IP thengoes through a public review and comment period. However, the technical study may have its own review period if it's published separately or if appropriate for a complex or controversial project The Regional Water Quality Section or field office takes the lead on advertising the report and collecting comments. The value of all these steps is that they help ensure transparency and a good public process. Concerns can be identified early in the process and addressed.

2.7 Implementation and Monitoring

The final step in assembling a TMDL package is the development of the water quality improvement plan (WQIP) which is part of the joint submittal prepared by EAP and WQ TMDL staff. Although Washington State considers it to be integral to the TMDL process, most of this portion of the WQIR/WQIP is currently not approved by EPA Region 10.

The WQIP is prepared by the regional TMDL lead and contains the best available actions neededto achieve the goals of the TMDL. This is typically done working in partnership with the advisory group. For nonpoint sources this may be very complex and take much of the TMDL regional lead's time, since it may involve a variety of sources, methods, authorities, and opportunities for improvement. For point sources, once WLAs are established, defining implementation should follow from the TMDL goals and standard permit management practices. If there are significant compliance challenges for point sources, permit managers will want to work closely with the regional lead to ensure consistent messaging to permittees and accurate descriptions of compliance schedules in the the WQIP.

One element of a TMDL required for EPA approval of a TMDL is the monitoring plan, which is located in the WQIP section of the joint WQIR/WQIP NPDES permits must specify monitoring requirement necessary to determine compliance with effluent limits (CWA section 401(a)(2), 40 CFR 122.44(i)). EAP is currently revising Ecology's Effectiveness Monitoring program for TMDLs, which will be a core element of the monitoring plan. Permittees may need to do additional monitoring to comply with WLAs.

Most of the permit managers' work in the TMDL involves setting WLAs, since the allocations depend on specific approaches to reducing loading. Implementation planning consists of documenting the activities and approaches that were identified in the allocation-setting process. Likewise, the implementation strategy for point sources should lead logically to a monitoring plan for permitted dischargers.

Challenges for permit managers at this stage of TMDL development may include:

- Deciding on the compliance schedule for loading reductions.
- Building any adaptive management approaches into the compliance schedule.
- Determining an efficient level of monitoring that is both effective and affordable.

The final WQIR/WQIP is sent to EPA for approval. Upon approval, permit managers will use the TMDL WLAs to establish permit requirements in NPDES permits. Continued collaboration between permit managers and the TMDL regional lead is important to ensure that the implementation plan in the TMDL submittal package is consistent with how NPDES permits will ultimately be revised to implement the TMDL.

2.8 Implementing TMDLs into NPDES Permits

Although some TMDLs never seem to go away, it's a significant milestone when a TMDL study, allocations, and implementation plan is packed into the WQIR/WQIPand sent to EPA. If the TMDL is relatively simple or if EPA has been closely involved, then EPA's approval is likely to be relatively routine. But for a complex or controversial TMDL, EPA may need additional time for review before approving the TMDL. In some situations EPA may return the TMDL for Ecology to revise and resubmit.

Once a TMDL is approved, the WLAs will need to be converted into permit limits. This occurs in several ways, depending on the type of permit:

- For an individual permit, new effluent limits will be set at the next cycle of permit renewal.
- For the most municipal separate storm sewer system (MS4) permits, TMDL requirements are included in an appendix at the next permit renewal. In the case of the WSDOT MS4 permit,

Ecology must modify that permit, or issue an administrative order, at least every 18 months in order to incorporate new TMDL-related requirements.

- For other general permits, special conditions for TMDLs can be added at the next permit renewal, or the facility with the TMDL limit may need to go under an individual permit.
- For any permit and discharge, if the need for implementing the WLA is urgent, an administrative order can be issued to implement permit conditions until the next permit renewal occurs.

The EPA web-based training modules provide some good general guidance on implementing WLAs in NPDES permits: https://www.epa.gov/npdes/tmdl-npdes-permits-training-user-guides.

For individual permits, the WLA must be converted into effluent limits, according to the procedures described in Chapters 6 and 7 of this manual and in EPA's Technical Support Document (TSD). For toxic pollutants, maximum daily and monthly average limits are required. Other averaging periods may be appropriate for bacteria, nutrients and temperature.

For general permits, special limits can be calculated for a facility with a WLA, using the same process as for individual permits to translate WLAs into effluent limits. However, for stormwater general permits a different approach is generally taken. Although stormwater permittees receive a numerical WLA in the TMDL, permits prescribe BMPs as effluent limitations to implement the WLAs, an approach first specified by EPA in 2002 (EPA, 2002) and expanded on in 2014 (EPA, 2014). This will generally be combined with monitoring of the parameters related to the TMDL impairment. For the MS4 permit, the Water Quality Program has developed a draft document: "Guidance for Translating TMDLs into MS4 Permit Requirements" (Ecology, 2011).

Other conditions may be written into NPDES permits to implement a TMDL. These could include:

- Compliance schedules
- Phased effluent limits
- Specific monitoring requirements
- Special studies or plans to be developed to address aspects of TMDL compliance

At this point in the TMDL process, the permit writer can return to the body of the *Permit Writer's Manual* for guidance.

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