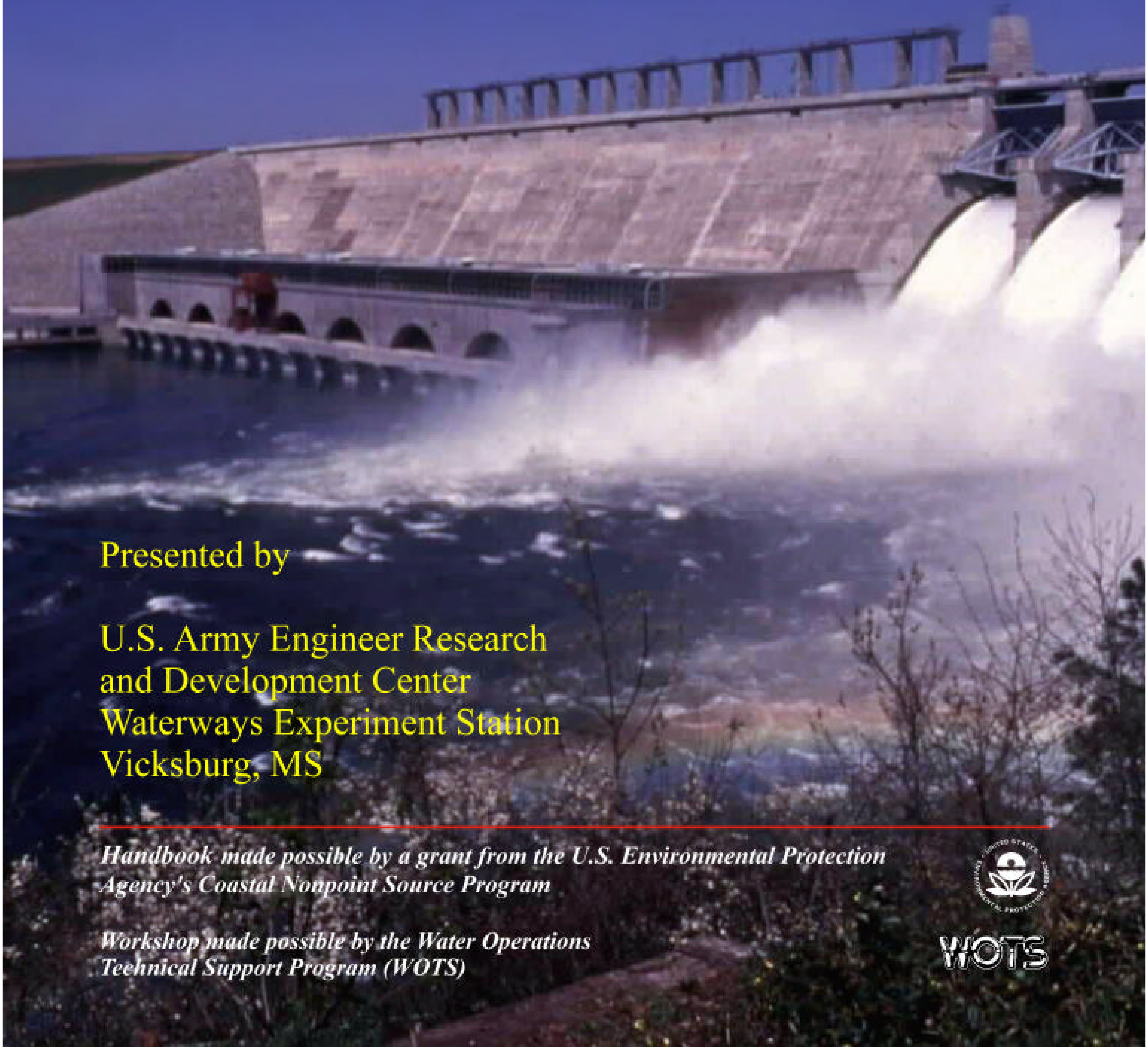
The WES Handbook on Water Quality Enhancement Techniques for Reservoirs and Tailwaters



Presented by

U.S. Army Engineer Research
and Development Center
Waterways Experiment Station
Vicksburg, MS

Handbook made possible by a grant from the U.S. Environmental Protection Agency's Coastal Nonpoint Source Program

Workshop made possible by the Water Operations Technical Support Program (WOTS)

THE WES HANDBOOK ON WATER QUALITY ENHANCEMENT TECHNIQUES FOR RESERVOIRS AND TAILWATERS

FORWARD

The WES Handbook on Water Quality Enhancement Techniques for Reservoirs and Tailwaters was developed under a grant from the US Environmental Protection Agency Office of Water - Nonpoint Source Pollution Control Program Division. Along with the handbook, several workshops were conducted to provide a forum for handbook evaluation and preliminary distribution. This effort was jointly shared by the Coastal and Hydraulics Laboratory (CHL) and the Environmental Laboratory (EL) at the Waterways Experiment Station (WES).

The handbook presents the thought process that a lake manager would follow to identify a specific water quality problem and associated processes, identify alternative solutions, and assess performance after an alternative has been installed. The handbook is divided into 5 areas:

watershed processes and basic limnology
data collection and analysis
problems and process identification
alternative enhancement techniques
performance assessment after installation

The handbook represents the efforts of numerous scientists and engineers from many organizations. The scope of the handbook is broad, but not exhaustive. Its publication in a 3-ring binder was purposeful to allow the easy insertion of additional technologies. The information was compiled by the workshop team of Steve Wilhelms, CHL; Steve Ashby and John Hains, EL; Steve Coffey, North Carolina Department of Environment and Natural Resources; Mark Mobley, Tennessee Valley Authority Engineering Laboratory (TVA), Dean Harshbarger, TVA, and Gary Hauser, TVA. Additional information was taken from “Water Quality Management for Reservoirs and Tailwaters: Report 1,” by G. Dennis Cooke and Robert H. Kennedy (1989) and “Water Quality Management for Reservoirs and Tailwaters: Report 2,” by Richard E. Price and Edward B. Meyer (1992). Steve Wilhelms and Steve Ashby were co-editors of the handbook. Special thanks go to Laurin Yates, CHL, who assembled the handbook and organized the workshops.

WES Handbook on

Water Quality Enhancement Techniques for Reservoirs and Tailwaters

Introduction

Most reservoirs experience some type of water quality problem ranging from contaminated inflow entering the reservoir to in-reservoir biological and chemical processes that degrade dissolved oxygen (DO). In-reservoir problems may be transferred to the tailwater by normal discharges through the reservoir's outlet structure. Even certain operations of the reservoir can create or exacerbate in-lake and tailwater problems.

In a high percentage of reservoirs, the underlying process causing many problems is called *eutrophication*. This is the addition of organic material and plant nutrients to the reservoir or lake that causes excessive production of algae and contributes to degradation of the oxygen. In extreme cases, the biological production is so great that the accompanying degradation (along with other in-water and bottom processes) results in complete use of the dissolved oxygen in the lower levels of the water column. Under these anoxic conditions, anaerobic biological and chemical processes cause a myriad of additional problems within and in reservoir releases.

To address these problems, a lake manager must have a reasonable knowledge of watershed processes, limnology, reservoir operations, enhancement techniques, and assessment methodologies. The objective of this handbook is to serve as a guide with which a lake manager could work through the process of identifying specific water quality problems, developing a sampling program to provide data for problem identification and alternative design, identifying an appropriate alternative, and assessing the performance of the alternative after installation. To support this process, Chapter 1 presents a brief review of watershed and limnological processes to provide an understanding of the sources and contributors to water quality problems. Chapter 2 covers data collection and analysis to identify and assess the extent of a problem. Chapter 3 examines a multitude of important issues that are often overlooked, including partnering with local government or organizations. Chapter 4 presents a review of many enhancement technologies. Chapter 5 presents an oft forgotten step of operation evaluation and adjustment.

We recognize that it is virtually impossible to exhaustively cover these topics in a single publication. Thus, this handbook is published in a binder so that its content is can be easily updated.

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WATERSHED - RESERVOIR PROCESSES: RELATION TO WATER QUALITY

1.1 OVERVIEW OF WATERSHED PROCESSES

Monitoring and modeling are tools to improve understanding of watershed processes. Monitoring provides empirical data for understanding important components. Modeling extends the knowledge gained from the empirical and provides a method of estimation or prediction of components. When combined, monitoring and modeling provide a picture of watershed processes that could not be possible with either tool in isolation.

1.1.1 WATERSHED HYDROLOGY

A useful daily watershed hydrology model is proposed by Arnold et al. (1998). The equation for the hydrology model is:

$$SW_t = SW + \sum_{i=1}^{n} (R_i - Q_i - ET_i - P_i - QR_i)$$

(summation I = 1 through n)

where SW is the soil water content minus the wilting point water content, t is the time in days, and R, Q, ET, P and QR are the precipitation, runoff, evapotranspiration, percolation and return flow, with units in mm.

1.1.1.1 Precipitation

Rainfall or snowmelt serve as precipitation for the daily hydrology model. Precipitation gauge data from the watershed or weather records from a nearby site are potential data sources. Methods for precipitation measurement and analysis of spatial variability in rainfall may be found in by Dunne and Leopold (1978) and Schwab et al. (1993). A stochastic weather generation model can also be used to simulate weather inputs. Precipitation statistics and monthly probabilities of receiving rainfall if the previous day was wet or dry are used in the weather generator developed by Nicks (1974) and Williams et al. (1984).

1.1.1.2 Evapotranspiration

Objectives and data requirements help in the selection of a method to estimate potential evapotranspiration (PET). The Priestly-Talyor (1972) method requires daily minimum and maximum air temperature and solar radiation. Ritchie (1972) outlines a procedure to compute soil and plant evaporation separately. The Penman-Monteith equation requires air temperature, solar radiation, wind speed, and relative humidity. Additional methods for PET estimation are available from Dunne and Leopold (1978) and Schwab et al.(1993).

1.1.1.3 Infiltration and Runoff

The SCS Curve Number Equation (CNE) (Ogrosky and Mockus, 1964; Mockus, 1972) may be used in determining runoff volume for source areas. The CNE may also be used in combination with the loading function for determining dissolved-phase nutrient loads in rural and urban runoff (Section 1.1.1.9 and 1.1.2.2-1.1.2.3).

The CNE is a research-based method for determining watershed runoff volume. The essential features of the equation are illustrated in Figure 1.1.1. SCS[1] (1986) provides guidance on use of the CNE.

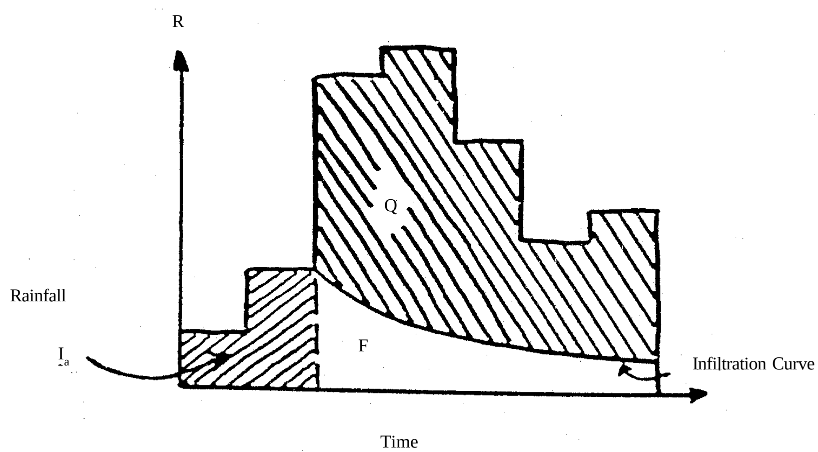

Figure 1.1.1. Relationship Between Precipitation, Runoff and Retention in the SCS Curve Number Equation (after McCuen, 1982)

CNE variables as defined by McCuen (1982)

- R = rainfall (in)
- I_a = initial abstraction, the precipitation volume which does not appear as runoff
- F = actual retention, the difference between precipitation volume and runoff volume
- Q = runoff (in) from source area k
- S = potential maximum retention

[1]Urban Hydrology for Small Watersheds (Technical Release 55, PB # 87-10158j0/AS A08) is available through:
National Technical Information Center
5285 Port Royal Road
Springfield, VA 22161
(703) 487-4650

The variables I_a and F represent evapotranspiration and ground water portions of the mass balance. Empirically I_a was found to be equal to 0.2 times the potential maximum retention. For a complete derivation of the CNE see McCuen (1982).

The CNE is given below to estimate runoff volume (Q) for each source area. A snowmelt term (M) (Haith and Tubbs, 1981) is added to the original CNE to yield:

$$Q = \frac{(R + M - 0.2S)^2}{(R + M + 0.8S)} \tag{1.1.1}$$

Where:

$$S = \frac{1000}{CN} - 10 \tag{1.1.2}$$

1.1.1.4 SCS Curve Number Determination

The magnitude of the CN is a function of the hydrologic condition of each source area. Characteristics that determine the CN are cover complex classification, soil group classification and antecedent moisture condition. Values of the CN for rural lands are summarized in Table 1.1.1. A complete listing of CNs along with guidance on their selection is given in (SCS, 1986).

Cover Complex. The cover complex determination is composed of a sequential assessment of cover type, treatment, and hydrologic condition.

a. **Cover type**
Cover type includes such land use categories as fallow, row crops, small grains, legumes, rotation meadow, pasture, brush, woods or farmsteads. For the first four cropping cover types, combinations of treatments describe the land use.

b. **Treatment**
Treatment aspect of the cover complex considers the percentage area covered with crop residue and the type of tillage system or combination.

c. **Hydrologic condition**
The hydrologic condition (Good, Fair or Poor) is based on the following factors affecting infiltration and runoff (SCS, 1986).
-density and canopy of vegetative areas
-amount of year-round cover
-amount of grass or close-seeded legumes in rotations
-percentage of residue cover on land surface (good$\geq$20%)

Factors improving average or better than average infiltration are considered to promote a good hydrologic condition. Poor hydrologic condition factors decrease infiltration and increase runoff.

Table 1.1.1. General SCS Runoff Curve Numbers for rural Lands Including Feed Lots

		Runoff Curve Number[2]			
Lane-Use	Surface-Condition	Soil Group A	Soil Group B	Soil Group C	Soil Group D
Fallow	0.22	77	86	91	94
Row Crop					
Straight Row	0.05	67	78	85	89
Contoured	0.29	65	75	82	86
Small Grain	0.29	63	74	82	85
Legumes or Rotation Meadow	0.29	58	72	81	85
Pasture[3]					
Poor	0.01	68	79	86	89
Fair	0.15	49	69	79	84
Good	0.22	39	61	74	80
Permanent Meadow	0.59	30	58	71	78
Woodland	0.29	36	60	73	79
Forest with Heavy Litter	0.59	25	55	70	77
Farmsteads	0.01	59	74	82	86
Urban (21%-27% Impervious Surfaces)	0.01	72	79	85	88
Grass Waterway	1.00	49	69	79	84
Water	0	100	100	100	100
Marsh	0	85	85	85	85
Animal Lot					
Unpaved		91	91	91	91
Paved		94	94	94	94
Roof Area		100	100	100	100

[1]Source: Young et al. (1982a)
[2]Source: US Department of Agriculture, Soil Conservation Service (1976). Values given are for Antecedent Moisture Condition II.
[3]Pasture should be considered "poor" if it is heavily grazed with no mulch. "Fair" pasture has between 50% and 75% plant cover and is moderately grazed. "Good" pasture is lightly grazed and has more than 75% plant cover.

Soil Group Classification. Most of the soils in the United States are classified by a hydrologic soil group letter A through D. The hydrologic soil group is determined by drainage class or infiltration rate. A list of U.S. Soils and their hydrologic group is given in SCS (1986). Local soil conservationists or soil scientists should be consulted if difficulty arises in determining soil groups.

Antecedent Moisture Condition. Once the hydrologic condition is compiled for a source area, the antecedent moisture condition (AMC) of the soil may be determined by summing the total precipitation in the 5 days prior to a storm. Soil moisture will affect infiltration capacity and the resultant runoff.

In most applications, the choice of average antecedent soil moisture condition (AMC II) is appropriate. AMC II CNs are used for input into the loading function model. During winter months, where soils are frozen AMC III may be appropriate for many areas. During dry conditions, for example, mid to late summer, AMC I may be appropriate. Careful consideration should be given in choosing an AMC for site and temporal conditions.

Curve numbers for AMC II are given in Table 1.1.1. The CNs for the other two antecedent conditions, AMC I and AMC III, are determined by conversion equations (Hawkins 1978). Direct runoff may be determined from Figure (1.1.2), as well as from equation (1.1.1).

1.1.1.5 Precipitation and Snowmelt

The design storm may be used to drive single event loading models. Precipitation events are highly variable infrequency, duration and intensity and are best characterized by statistical analysis of long-term precipitation records. From historical data analysis, precipitation intensity-duration curves are constructed. From the resulting probabilistic curves, the rainfall depth for a specified duration and recurrence interval for the design storm may be determined.

The design storm is defined as an estimate of rainfall depth for a specified duration (10 min-24 hours) for a specified return period (2-100 years). A probability distribution derived from the historic rainfall record is used to statistically estimate the design storm rainfall depth. The relationship between return period and probability of design storm event occurrence is defined by the following relationship.

$$T = \frac{1}{P} \tag{1.1.3}$$

Where:

T = return period in years
P = probability (0-1) an annual maximum event of any year will equal or exceed some given value

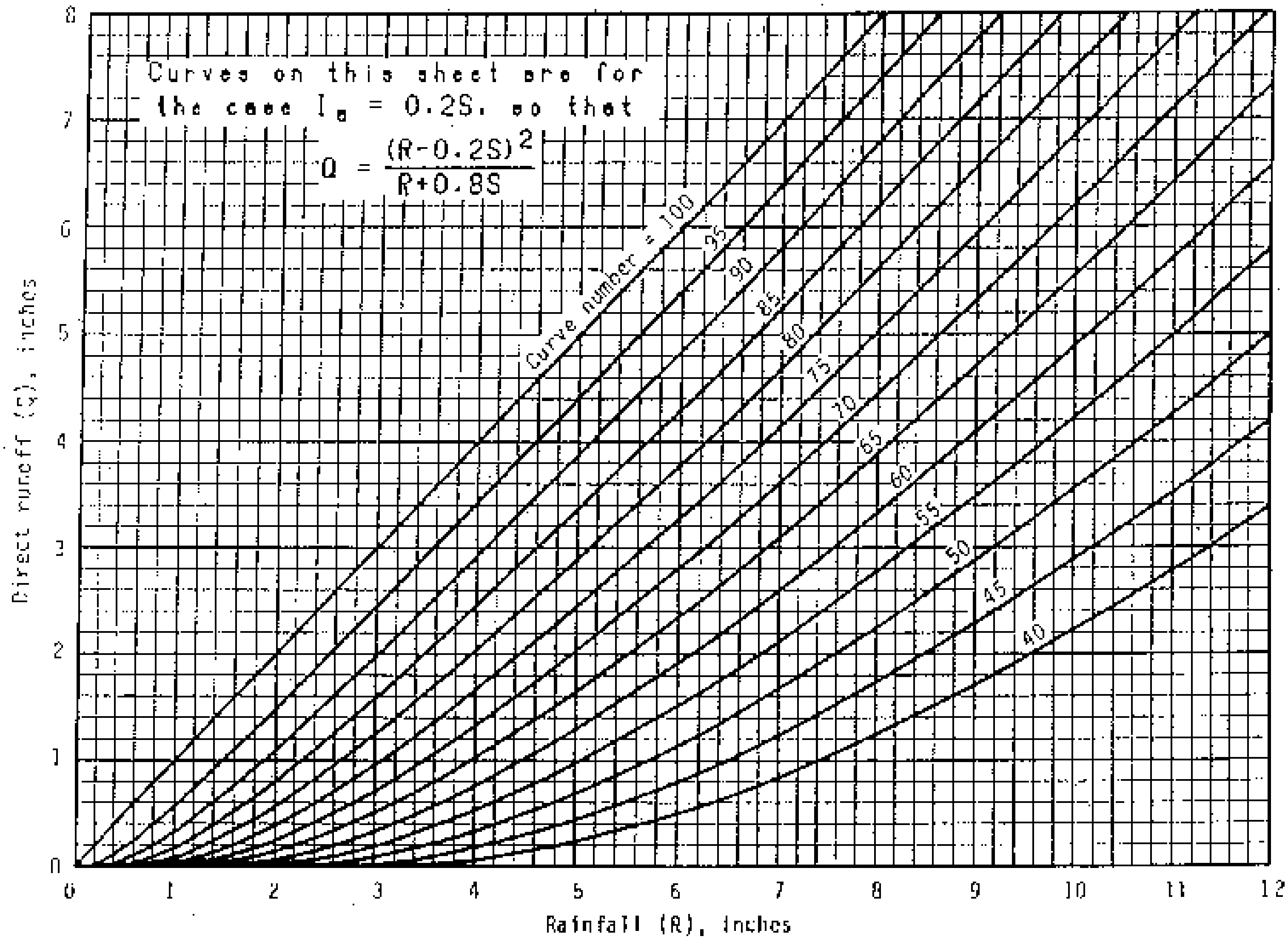

Figure 1.1.2. Graphical Solution to the SCS Curve Number Equation (after SCS, 1986)

The following examples illustrate the important concepts useful for applying the design storm.

The interpolated 2-year, 24-hour rainfall depth for Raleigh, North Carolina, is 3.6 in. (Hershfield, 1961; SCS, 1986). Substituting the return period of two years and rearranging equation 1.1.3 yields the following probability.

$$P = \frac{1}{2} = 0.5 \tag{1.1.4}$$

There is a 50% probability that a storm of 3.6 in or greater will occur in any year. Similarly, the annual probabilities of the 10-year and 25-year design storms are 0.1 and 0.04.

The maps contained in the rainfall frequency atlas (Hershfield, 1961 and SCS, 1986) were obtained from smoothing, translating and interpolating rainfall data at point locations. Rainfall depths obtained from the atlas and other similar sources should only be regarded as representative values and uncertainties or errors are associated with each point estimate. Tung (1987) quantified the rainfall estimate uncertainty procedures for estimating the mean and coefficient of variation of design storm rainfall depth for 1-100 year return periods and 30 minute to 24 hour storm durations.

We suggest the 2-year, 10-year and 25-year design storms as a range of precipitation inputs which is preferred over a single point estimate for an event-based assessment. The use of a range provides an indication of the variability in extreme events. The 2-year design storm is the maximum expected precipitation value for an average year. The 10-year design storm provides an intermediate rainfall and shares the same return period as critical stream low flows (7_Q10). We also recommend the (25-year design storm as suggested by Young et al., 1982) for evaluating nonpoint source runoff as in the case of feedlots. Maps providing rainfall depths for the 48 contiguous states for the 2-year, 10-year, and 25-year design storms are available from (SCS, 1986; Hershfield, 1961). We suggest examining weather records to determine average storm duration during the season of interest.

Snowmelt estimation is essential to evaluating the water balance and may transport a significant portion of pollutant runoff in northern temperate regions. Modeling snowmelt is difficult since many natural processes affect the snowpack energy balance including solar radiation and the changes in sensible heat flux that vary with wind speed. Factors affecting the energy balance include radiation exchanges at the snow surface, sensible heat exchanges with the overlaying air, condensation, evaporation and freezing in addition to the melting phase changes of snow.

Hendrik et al. (1971) developed a snowpack energy balance model that features a daily time-step and requires a large set of site specific daily weather data. Spatial Variability in snowmelt is represented by selecting topographically uniform snowmelt zones with respect to slope and directional aspect (north to south) that are assumed to produce similar snowmelt runoff. Model terms include slope aspect, vegetative cover coefficient, decimal fraction of daily cloud cover, clear sky daily solar radiation, maximum daily temperature, mean daily wind speed, mean daily dew point and daily rainfall.

Snowmelt. (M) as given in GWLF can also be estimated from the degree-day equation given by Haith (1985) from daily precipitation depth and mean daily temperature. While this approach is not expected to provide the most reliable results, some snowmelt estimation is required for many northern temperate regions. Precipitation is assumed to be snow when:

$$T_t \leq 0\ ^{o}C \tag{1.1.5}$$

Snowmelt on day t may be estimated by the degree day equation proposed by Haith (1985). For daily mean air temperature, T, (^{o}C) greater than zero, then

$$M = 0.45\, T_t \text{ for } T > 0 \tag{1.1.6}$$

A degree-day model is based solely on average daily temperature effects on snowmelt. Thus, a degree day equation may not adequately represent the day to day variability in snowmelt.

1.1.1.6 Functional Relationship of Loads and Concentrations

For the purpose of watershed management, pollutant loads in kilograms per time period will be used to evaluate the contribution that each source makes to the receiving water body. Ranking of pollutant sources based on total load contribution is useful for assessment and targeting source areas for treatment.

For eutrophication modeling and assessing pollutant impact on the ecosystem, the most useful quantification of pollutant flux to the water body is the nutrient concentration. The stream and lake models (Section 4.0) require average period of assessment pollutant concentrations.

1.1.1.7 Hydrologic Budget for the Loading Function

The GWLF model (Haith and Shoemaker, 1987) features a simple, daily time step hydrologic budget for the lumped parameter model. The daily water budget is summarized in Figure 1.1.3.

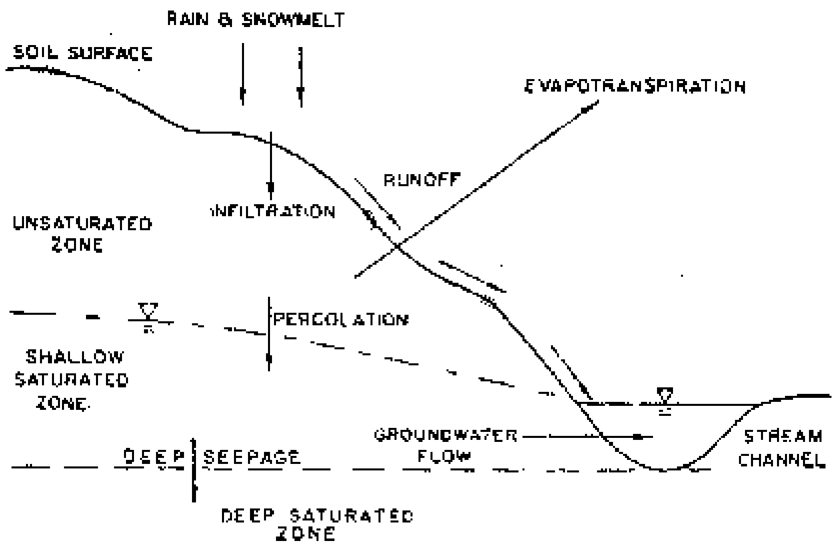

Figure 1.1.3. Water Budget for the Lumped Parameter GWLF Stream Flow Model.
(after Haith and Shoemaker, 1987)

The daily unsaturated zone and shallow saturated zone water budget for day t, in moisture units of centimeters (cm) is:

$$U_{t+1} = U_t + R_t + M_t - Q_t - E_t - Pc_t \tag{1.1.7}$$

$$S_{t+1} = S_t + PC_t - G_t - Dt \tag{1.1.8}$$

Where:

U_{t+1} = unsaturated zone soil moisture
S_{t+1} = shallow saturated zone soil moisture
R_t = rainfall
M_t = snowmelt
Q_t = watershed runoff Q_{kt} (for all k)
E_t = evapotranspiration
Pc_t = percolation into the shallow saturated zone
G_t = ground water flow or discharge into the stream
D_t = deep seepage flow to the deep saturated zone

Evapotranspiration. (E_t) for day t may be estimated by the following equation:

$$E_t = Cv_t \ Pe_t \tag{1.1.9}$$

Where:

E_t = evapotranspiration (cm)
Cv_t = monthly cover coefficient (dimensionless)

Literature sources for monthly cropland cover coefficients are Chow (1964) and Davis and Sorensen (1969).

Potential evapotranspiration is estimated in the GWLF model by the Hamon (1961) equation. The model calculates saturated vapor pressure. The model user is required to supply monthly average daylight hours (Mills et al., 1985) and mean daily temperature.

Percolation. Percolation (Pc_t) into the shallow saturated zone occurs when unsaturated zone soil moisture exceeds field capacity. Percolation is calculated by normalizing the unsaturated zone to represent the soil moisture content at field capacity. For the unsaturated zone at field capacity $U_t = 0$ (Haith and Shoemaker, 1987). The following mass balance equation estimates percolation for day t:

$$Pc_t = Max\ (O;\ U_t + R_t + M_t - Q_t - E_t) \tag{1.1.10}$$

Ground water discharge (G_t) (Figure 1.1.3) to the stream channel and deep seepage movement of water from the shallow saturated zone to the deep saturated zone is modeled by a simple linear reservoir (Haan, 1972):

$$G_t = r\ S_t \tag{1.1.11}$$

Where:

G_t = ground water discharge (cm)
r = ground water recession constant (dimensionless)
S_t = soil moisture of shallow saturated zone (cm)

The user must supply a ground water recession constant r, which may be estimated from the hydrograph separation procedure (Chow, 1964). From a partial stream flow record of the planning watershed a hydrograph is plotted on semilogarithmic paper to determine the separation between surface and subsurface runoff.

Deep Seepage. Deep seepage (D_t) may be estimated from the residual of ground water discharge. A conservative approach assumes deep seepage is negligible. The conservative approach assumes deep seepage is negligible. The conservative mass balance, allowing no deep seepage, assumes all watershed precipitation exists as evapotranspiration or stream flow (Haith and Shoemaker, 1987):

$$D_t \;=\; s\,S_t \qquad (1.1.12)$$

Where:

D_t = deep seepage (cm)
s = rate constant for deep seepage (dimensionless)
S_t = soil moisture of the shallow saturated zone (cm)

1.1.18 Nutrient Budget for the Loading Function

Screening-level methods for estimating source area nutrient loads include the export coefficient (unit area load) and the loading function. Reckhow et al. (1980) developed a procedure for use of nutrient export coefficients with an empirical lake model for phosphorus. The procedure included an analysis of pollutant loading uncertainty expressed as a high and low pollutant export range associated with the most likely export. Reckhow et al. 1980 lists phosphorus and nitrogen export coefficients from well-designed runoff studies for the major land uses, along with size of study area, annual precipitation and soil characteristics. The following nonpoint nutrient sources are not included in the discussion, but references are noted for each source:

Nutrient Sources	References
atmospheric inputs	Reckhow and Chapra, 1983
septic tank inputs	
nutrient regeneration from water body sediments	
waterfowl fecal matter	Manny et al., 1975

To estimate the total watershed pollutant load we suggest the use of the nutrient budget. Reckhow et al. (1988) provide guidance on nutrient budget estimation for state-level applications. The methods provided in this methodology include loading functions, export coefficients and the references provided herein.

Both annual and seasonal variability in pollutant loading are important to the assessment and they should be quantified. Annual variability is generally high so that several modeling years may be required to achieve a satisfactory estimate of the mean annual pollutant export. Seasonal land use changes within an agricultural watershed may be modeled using the seasonal period of assessment with monthly loading estimates.

The generalized watershed loading function (GWLF) model developed by Haith and Shoemaker (1987) is the basis for the following nutrient and sediment assessment. The GWLF procedure produces monthly estimates of sediment, phosphorus and nitrogen loads based on a daily time step for computing runoff and ground water discharge. A computerized version is available from the first author. Monthly nutrient loads in stream flow are determined by summing the following (mass balance) components of the dissolved and solid-phase watershed nutrient loads in kilograms, for month m.

$$LD_m = DP_m + DR_m + DG_m + Du_m \tag{1.1.13}$$

$$LS_m = SP_m + SR_m + SU_m \tag{1.1.14}$$

Where:

LD_m = total dissolved nutrient mass
DP_m = dissolved point source nutrient mass
DR_m = dissolved rural nutrient mass
DG_m = dissolved ground water nutrient mass
DU_m = dissolved urban nutrient mass
LS_m = total solid-phase nutrient mass
SP_m = solid-phase point source nutrient mass
SR_m = solid-phase rural nutrient mass
SU_m = solid-phase urban nutrient mass[2]

1.1.1.9 Cropland and Forest Dissolved Nutrient Loading

We describe the dissolved nutrient loading component of the GWLF (Haith and Shoemaker, 1987) model for rural source areas. The model is based on the CNE for runoff estimation and earlier applications of the dissolved loading function are documented by Delwiche and Haith (1983) and Haith and Tubbs (1981). The event based model applies to agricultural, forest, and construction land use pollutant runoff.

The watershed is divided into source areas based on soil type, vegetative cover, management practice and hydrologic response. Each source area is assumed to make an independent load contribution to the watershed outlet. Nutrient export for each source area is summed over the precipitation event. Loads area then transported to the receiving watercourse in runoff. The method assumes a conservative transport mechanism allowing no attenuation of nutrients from "edge-of-field" to the watershed outlet (Haith and Tubbs, 1981).

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[2]Dr. Douglas A. Haith, Department of Agricultural Engineering, Riley-Robb Hall, Ithaca, NY 14853, 607-255-2802

Dissolved nutrient load (kg) from source area k, for precipitation or snowmelt on day t, is given by:

$$LD_{kt} = 0.1\ Cd_{kt}\ Q_{kt}\ A_k \tag{1.1.15}$$

Where:

Cd_{kt} = dissolved nutrient concentration (mg/l)
Q_{kt} = runoff 9cm), calculated by CNE (Section 1.1.1.3)
A_k = source area (ha)
0.1 is a units conversion constant

Dissolved Nutrient Concentrations. Dissolved nutrient concentrations ($Cd_{kt)}$ in runoff vary primarily due to crop type. Secondary considerations include animal waste and manufactured fertilizer input, soil type, and the timing of precipitation events (Haith and Tubbs, 1981). Local dissolved nutrient flow-weighted mean concentration or event mean concentration edge-of-field data may be available for local modeling applications. Use of local data is preferred over the selection of default values (Table 1.1.2).

Table 1.1.2. Dissolved Flow-weighted Mean Concentration Nutrients in Agricultural Runoff.

	Nitrogen	Phosphorus
Land Use	(mg/1)	(mg/1)
Fallow[1]	2.6	0.10
Corn[1]	2.9	0.26
Small Grains[1]	1.8	0.30
Hay[1]	2.8	0.15
Pasture	13.0	0.25
Barn Yards[2]	29.3	5.10
Forest	0.3[3]	0.0007[4]

[1]Dornbursh, et al. (1974)
[2]Edwards, et al. (1972)
[3]Johnson (1975)
[4]Omernik (1977)
(after Haith and Shoemaker, 1987)

Gilbertson provides additional default Cd values for incorporated and unincorporated manure and rainfall and snowmelt runoff in Table 1.1.3.

More detailed infiltration equations include the Horton equation (Schwab et al., 1993) and the method proposed by Green and Ampt (1911).

1.1.2 EROSION AND SEDIMENTATION

1.1.2.1 Rural Sediment and Sediment-Attached Nutrient Loading

We describe the sediment and sediment-attached nutrient loading component of the GWLF (Haith and Shoemaker, 1987) model. The procedure is based on the erosion estimates from the Universal Soil Loss Equation (USLE) (Wischmeyer and Smith, 1978). The method for

Table 1.1.3. Estimated Concentration of Total Nitrogen, Total Phosphorus, and Chemical Oxygen Demand Dissolved in Runoff from Land With and Without Livestock or Poultry Manure Surface Applied[1] at Agronomic Rates.

	Rainfall runoff						Snowmelt runoff		
Cropping condition	Total N		Total P		COD		Total N	Total P	COD
	Manure		Manure		Manure				
	With	Without	With	Without	With	Without	With Manure		
	Parts per million								
Grass	11.9	3.2	3.0	0.44	360	50	36	8.7	370
Small grain	16.0	3.2	4.0	0.40	170	20	25	5.0	270
Row crop	7.1	3.0	1.7	0.40	88	55	12.2	1.9	170
Rough plow	13.2	3.0	1.7	0.20	88	55	12.2	1.9	170

[1] Incorporating manure in the soil would result in element concentrations the same as those for “without manure.”

(after Gilbertson et al., 1979)

quantifying sediment-attached pollutant loads from watershed source area to a receiving water body is:

1. Estimate soil loss from a unit source area using the USLE.

2. Estimate sediment delivery to receiving water body.

3. For event-based sediment attached nutrient loads multiply sediment delivered by the concentration of nutrient in the eroded soil (sediment) as in equation 1.1.20.

Soil Loss. Haith (1985) gives an event-based version of the USLE for soil loss on day t:

$$Xt = 1.29 \qquad A\ K\ (LS)\ C\ P \qquad (1.1.16)$$

Where:

X_t = soil loss generated (mg)
A = area of erosion source (ha)
K = soil erodibility
LS = topographic factor
C = cropping factor
P = practice factor
K, LS, C, and P are dimensionless
1.29 is a units conversion constant

Rainfall Erosivity. E is a measure of the energy of rainfall and runoff (including snowmelt) that causes erosion. Rainfall erosivity can be computed from rainfall intensities based on short term records (15 min or hourly), however, this data intensive approach is seldom practical.

Alternatively, erosivities may also be obtained from daily rainfall data by using a regression equation developed by Richardson et al. (1983):

$$E = 6.46a\ (R^{1.18}) \qquad (1.1.17)$$

Where:

E = daily rainfall erosivity
R = daily rainfall (cm)
a = coefficient that varies with location and season (dimensionless)

Richardson et al. (1983) have determined cool season (October-March) and warm season (April-September) values for coefficient a for 11 locations in the eastern United States. Although regression equations such as equation 1.1.17 are generally convenient predictive models, their usefulness depends on the confidence intervals for the model development data set. Neither Richardson et al. (1983) nor Mills et al. (1985) specifically present confidence intervals. However, Richardson’s erosion index versus rainfall distribution indicate that the confidence in this case may be fairly low (see Figure 1.1.4.). However, it is probably the best available indicator of the rainfall erosion relationship.

Soil Erodibility. Soil erodibility K) indicates the susceptibility of soil to erosion and is a function of soil type. K values are available from local soil and water conservation districts, state offices of the Soil Conservation Service (SCS), and from Stewart et al. (1975).

Topographic Factor. The topographic factor (LS) combines source area slope length and steepness into a single factor for estimating soil erosion by water. Wischmeier and Smith (1978) provide a table, figure and an English units equation for determination of source area LS. Data for computing LS are available from topographic maps or county soil surveys.

Cover and Management Factor. The cover and management factor (C) describes the protection of the soil surface by plant canopy, crop residues, mulches, etc. The maximum C value is 1.0, corresponding to fallow cropland with no protection. Cropland C values change substantially during the year in response to seasonal activities such as crop planting, tillage, growth, and harvest. Cover and management factor values are available for each crop stage in different management systems (Wischmeier and Smith, 1978) and Mills et al. (1985). The average annual values from Stewart et al. (1975) are adequate for an initial approximation screening study. Wischmeier and Smith (1978) have also developed C factors for construction

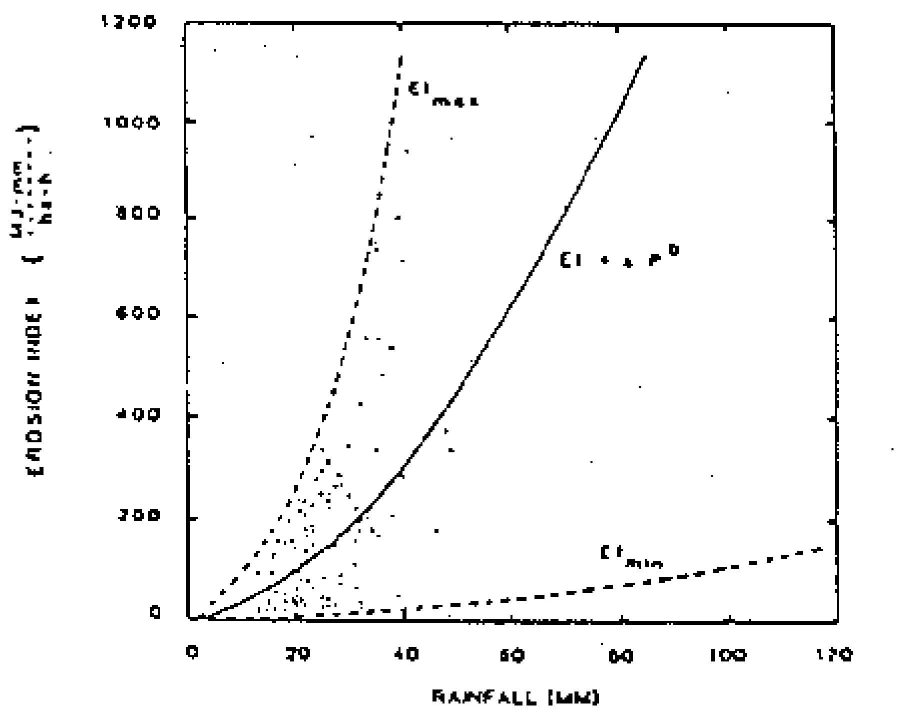


Figure 1.1.4. General Relationship Between Rainfall Amount and Erosion Index for Individual Storms. (after Richardson et al., 1983).

sites; pasture, range and idle land; undisturbed forests; and mechanically prepared woodland sites.

Supporting Practice Factor. The supporting practice factor (P) is a measure of the effect of soil conservation practices such as contouring and terracing on cropland erosion. Stewart et al. (1975)

provide P values for agricultural practices, the maximum value of which is 1.0. On construction sites, however, values for P may be greater than 1.0. The New Jersey sediment control manual (NJ SSCC, 1982) provides P values for these conditions.

Sediment Delivery. The movement of sediment and sediment-attached nutrients for a given runoff event, from source area to receiving water body is a complex process. Over the course of many events pollutants may be deposited, resuspended, scrubbed from solution by adsorption, undergo chemical reaction or biological uptake and redissolved by desorption or decay. In spite of sediment yield process complexities, it is possible to focus on important factors and derive an approximate estimate of pollutant delivery.

According to Mills et al. (1985), annual watershed sediment yield due to surface erosion as:

$$Y = SD \sum_{k} X_k A_k \qquad (1.1.18)$$

Where:

Y = average annual sediment yield (tons/yr)
X_k = average annual erosion from source area k as given by the USLe (t/ha)
A_k = area of source area k (ha)
SD = watershed sediment delivery ration (dimensionless)

The sediment delivery ratio (SD) is the mass of sediment delivered to the point of measurement divided by the mass of soil loss due to gross erosion. All of the soil that is eroded or transported may not reach the point of measurement. Many sediment delivery models have been developed using a variety of geomorphic terms to calculate the sediment delivery ratio. Criteria for model selection should consider source area type (individual field, subwatershed or watershed), geographic location and sediment traps in the watershed.

For individual fields, the distance from the source area to the watercourse may be used to estimate sediment delivery. Reckhow et al., 1988 developed a distance-based SD from Texas, Oklahoma and extreme southern Kansas). The distance-based SD for either an individual field or watershed is:

$$1n \ (SD) = 1.10 - 0.34 \ 1n \ (D) \qquad (1.1.19)$$

where:

$R^2 = 0.8$
S.E. = 0.356
n = 25

1n = the base e logarithm

For an individual field:

D = distance (m) from the center of the land parcel along the down slope gradient to the nearest surface water body

For a subwatershed or watershed:

D = one half the length of the watershed measure along the mainstem stream

A area-based SD (Roehl, 1961) is:

$$SD = 0.47A^{-1.25} \qquad (1.1.20)$$

Where:

A = watershed area in km^2.

The scatterplot of the original data to derive equation 1.1.19 is from southeastern U.S. watersheds (Figure 1.1.5). Modelers should consider the most likely SD and chose a range of high and low SDs based on data variability.

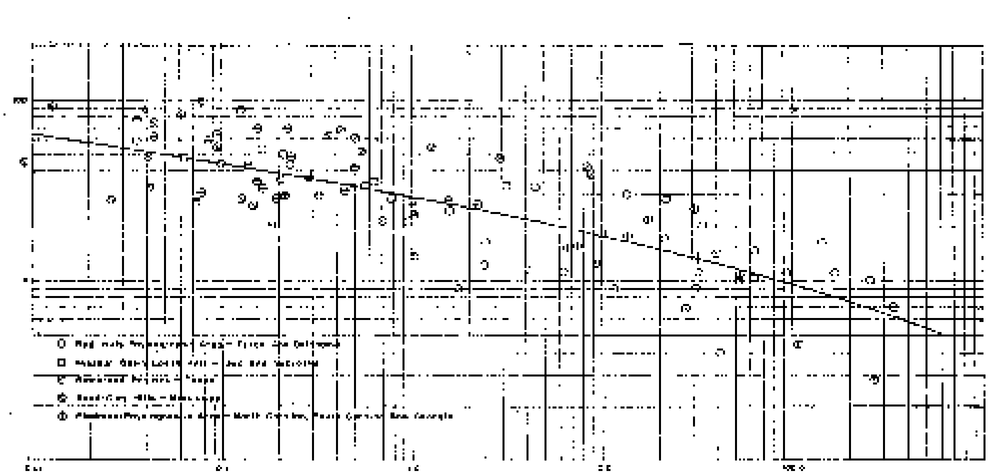


Figure 1.1.5 Sediment Delivery Ration as a Function of Watershed Drainage Area. (after Roehl, 1962)

Haith et al. (1985) have observed that large watershed sediment yields often do not coincide with major erosion periods. For example, in the eastern United States, most soil erosion is caused by late spring and summer intense rainstorms, but most sediment discharge occurs during late winter and early spring runoff. They reason that runoff during erosive periods is often insufficient to transport eroded soil far from a source area. Subsequent large events are assumed to flush portions of the accumulated sediment from the watershed drainage network to the receiving water body. Haith et al. (1985) propose a “sediment year” for the Eastern United States running fro April through March.

Although general procedures are not available for estimating seasonal sediment yields, Haith et al. (1984) achieved “satisfactory” results in an 850 km^2 New York watershed using monthly approximations. They assumed sediment yield during a given month m, Y_m , to be proportional to $Q_m^{1.2}$ where Q_m is the watershed runoff during month m. Y_m is then derived from historical values for average annual sediment yield Y as obtained from equation 4-18. A seasonal yield is the sum of the Y_m for the months in a given season.

Sediment-Attached Nutrient Loading. The loading function for a sediment-attached nutrient runoff (Haith and Tubbs, 1981) from a source area k, at time t, is:

$$LS_{kt} = 0.001\ Cs_{kt}\ X_{kt}\ SD_k \text{ and } A_k \qquad (1.1.22)$$

Where:

= solid-phase nutrient load (kg)
= concentration of the nutrient in sediment (mg/kg)
= soil loss (t/ha) (from the USLE)
= sediment delivery ratio (dimensionless)
= area (ha)
0.001 is a units conversion constant

The concentration Cs may be determined by direct analysis of eroded soil (sediment) or in situ soil samples. Since erosion is selective for smaller soil particles, in situ concentrations must be adjusted by an enrichment ration, thus:

$$Cs = en\ Ci \qquad (1.1.21)$$

Where:

cs = nutrient concentration in sediment (mg/1)
en = enrichment ration (dimensionless)
ci = nutrient concentration in situ soil (mg/1)

Mills et al. (1985) discuss methods for arriving at values for en. However, Haith and Tubbs (1981) maintain that enrichment ratios for nutrients typically vary from 1 to 4 and are essentially unpredictable.

Since they found that most reported values are at the lower end of the 1-4 range, they assumed that the concentration of nutrients in sediment reaching a stream is approximately twice that in the in situ soil.

1.1.2.2 Feedlot Pollutant Runoff

Young et al. (1982) developed a pollutant runoff model for estimating total phosphorus, chemical oxygen demand (COD) and biological oxygen demand (BOD) loads from animal feedlots for single events. A submodel of the AGNPS model for estimating total nitrogen runoff is also incorporated in to the feedlot model given below (Young et al., 1987). As with the dissolved nutrient loading function for cropland and forest (Section 1.1.9) pollutant loads in the feedlot model are determined by combining site-specific or default pollutant concentrations with runoff volume estimates from the SCS Curve Number Equation. Animal waste pollutant load attenuation from the feedlot edge to receiving watercourse is estimated by regression models available for several buffer area cover types.

The feedlot model (Young et al., 1982; Young et al., 1987) was developed for estimating pollutant runoff for Minnesota feedlots, but the model is applicable to other locations. The BOD assessment is based on a COD/BOD ration conversions. Of the nutrient runoff models, the primary author has more confidence in the phosphorus runoff estimates than the nitrogen runoff estimates due to the complexities of nitrogen adsorption and desorption (Robert A. Young, personal communication). Planners should therefore consider the increased uncertainty associated with the nitrogen runoff estimate compared to the phosphorus runoff estimate. A computerized version of the feedlot model for estimating phosphorus and COD runoff is available.[3]

To begin estimating feedlot pollutant runoff, name receiving water body and uniquely identify the feedlot. Then prepare a feedlot site plan and calculate the acreage of feedlot watershed subareas described below and shown in Figure 1.1.6.

[3]Dr. Robert A. Young
USDA-ARS
North Central Soil Conservation Research Laboratory
Morris, MN 56267
612-589-3411

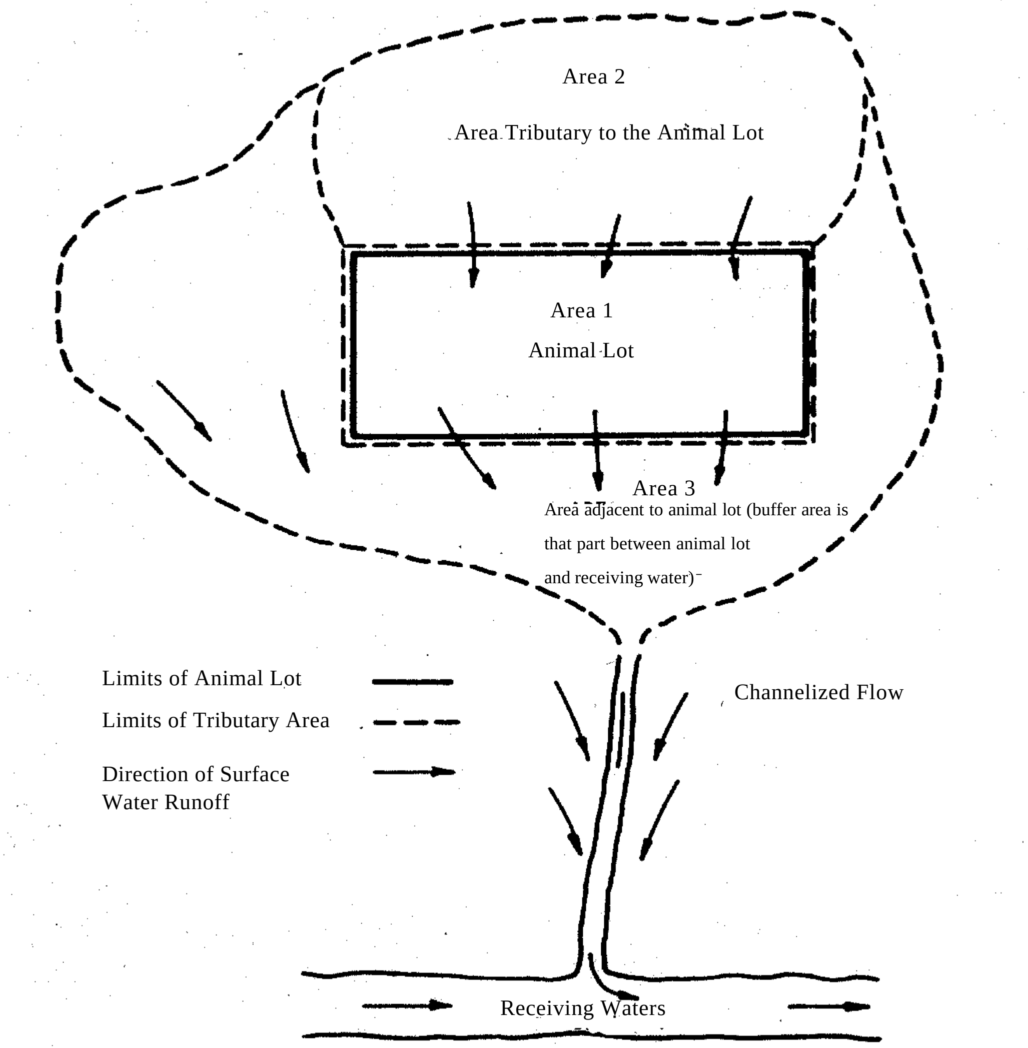


Figure 1.1.6. Feedlot site plan. (After Young et al. 1982)

Area 1 Animal lot: confined animal area, devoid of vegetation, subtract area covered by roofing.

Area 2 Tributary area: roof, upslope and tributary areas contributing runoff to Area 1. Subareas (2a, 2b, and so on) with different cover types or soils may be distinguished within Area 2. Area 2r is the roof area.

Area 3 Adjacent area: includes areas adjacent or lateral to Areas 1 and 2 that contribute to the discharge point. Subareas within Area 3 (3a, 3b, and so on) may be distinguished as in Area 2.

Buffer area: area downslope of feedlot between feedlot and discharge point, includes grassed waterway.

Discharge point: The discharge point (Figure 1.1.6.) Is the location closest to the feedlot where runoff changes from sheet flow to channelized flow. When runoff volume becomes channelized, no further pollutant attenuation occurs from vegetative filtration. The discharge point may be physically defined as a tile inlet, gully or rill. If a grass waterway is used for feedlot runoff, treatment or discharge, it should be included as part of the buffer. The discharge point is the end of the effective treatment point or end of the grass waterway (Young et al., 1982).

Channelized flow area: ditch, gully, or rill

Determine the CN for animal lot Area 1, from Table 1.1.4 Curve Numbers for Areas 2 and 3 can be determined from Table 1.1.1.

Define feedlot animal types and determine pollutant ratios from Table 1.1.4.

The pollutant concentration (C_1) for Area 1 feedlot animal waste may be determined by the following procedure.

Determine animal unit density (AUD) :

$$AUD = EAU/A \text{ (if } AUD > 100\text{, use 100)} \tag{1.1.23}$$

Where:

EAU = equivalent animal units
= Number of animals X pollutant ratio
A = area of feedlot area (acres)

Table 1.1.4 Ratio of COD, P, and N produced by various animals to that produced by a 1,000 pound slaughter steer.				
Animal Type	Design weight[a]	Pollutant Ratios		
		COD	P	N[b]
	Pounds			
Slaughter Steer	1,000	1.00	1.00	1.00
Young Beef	500	0.50	0.51	0.60
Dairy Cow	1,400	1.96	0.92	1.68
Young Dairy Stock	500	0.70	0.33	0.46
Swine	200	0.17	0.27	0.26
Feeder Pig	50	0.04	0.07	0.07
Sheep	100	0.18	0.06	0.13
Turkey	10	0.02	0.03	0.02
Chicken	4	0.01	0.01	0.01
Duck	4	0.01	0.01	0.01
Horse	1,000	0.42	0.42	0.81

[a]Interpolation of values should be based on the maximum weight animals would be expected to reach.
[b]Data from Midwest Plan Service (1975) except for swine, which is from American Society of Agricultural

Determine percent manure pack:

In areas of high animal unit density (AUD) (Young et al., 1982), almost all of the rainfall and runoff in and from the lot comes in contact with animal wastes before leaving the lot. Where AUD is low, some of the runoff may escape contact with animal waste and thus contain no fecal contamination. Sweeten and Reddell (1976) assumed uniform spreading of cattle manure over a feedlot area over a given time period to estimate percent manure pack. If one beef animal (1,000 lbs) covered approximately 0.001 acres per day, assuming animal waste scraping or waste removal (cleaning) frequency of 10 days, an AUD of 100 head of beef cattle per acre will result in 100 percent manure pack, Figure 1.1.7. Where feedlot systems include outdoor exercise areas, AUDs should be adjusted for the percent time spent in these areas. P, in percent manure pack below 100, from 85, 4500 and 300 mg/1 respectively (Young et al., 1982, 1987).

Determine runoff volume from Area 2 (V_2 acre-in) by the CNE (Section 1.1.3.) by summing runoff volumes from each subarea.

For Area 2, if runoff volume (V_2) is greater than 30 acre-inches, the pollutant concentration (C_2) in mg/1 at the feedlot edge is calculated by:

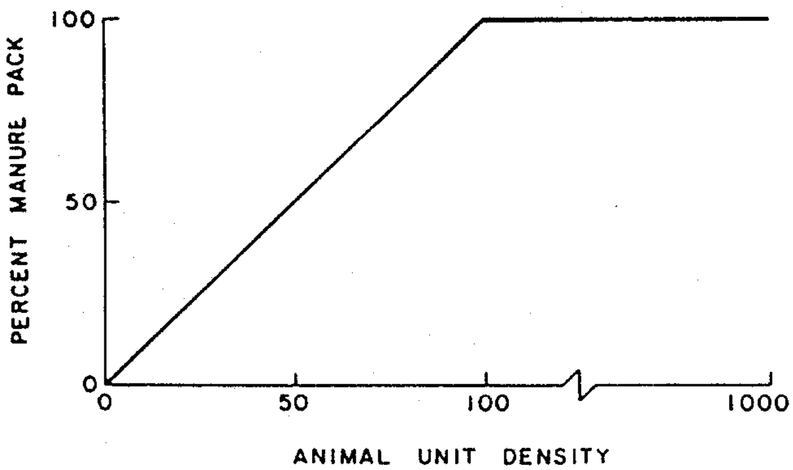


Figure 1.1.7. Percent manure pack vs. animal unit density.
(from Young et al., 1982)

$$[\text{percent manure pack} \times 45 \times (V_1 + 30) + 60\,(V_2 - 30)\,] / (V_1 + V_2) = COD \qquad (1.1.24)$$

$$[\text{percent manure pack} \times 0.85 \times (V_1 + 30) + 2\,(V_2 - 30)\,] / (V_1 + V_2) = P \qquad (1.1.25)$$

$$[\text{percent manure pack} \times 3 \times (V_1 + 30) + 12\,(V_2 - 30)\,] / (V_1 + V_2) = N \qquad (1.1.26)$$

<u>Determine pollutant concentration at feedlot edge</u> (C_F) by adding runoff water and pollutants from Area 1 and Area 2:

$$C_F V_F = C_1 V_1 + C_2 V_2 \qquad (1.1.27)$$

Where:

C_F = concentration of runoff at the feedlot edge (mg/1)
V_F = volume of runoff at feedlot edge (ac-in)
C_1 = concentration of runoff in the feedlot (mg/1)
V_1 = volume of runoff from feedlot (ac-in)
C_2 = concentration of runoff from Area 2 (mg/1)
V_2 = volume of runoff from Area 2 (ac-in)

Default values for Area 2 and Area 3 background concentrations for P, COD, and N are provided in Table 1.1.5 and Table 1.1.6

Table 1.1.5. Background Runoff Concentrations of Total Phosphorus and Chemical Oxygen Demand For Rural Land Uses.		
Source	P (mg/l)	COD (mg/l)
Native Prairie[1]	0.2	49
Corn[2]	0.9	
Continuous Corn[3]	1.1	
Native Grass[4]	0.1	31
Wheat[5]	1.4	
Pasture[5]	1.0	
Alfalfa[5]	2.1	
Corn[6]	0.3	
Soybeans[6]	0.5	
Small Grain[6]	0.6	
Cropped Watershed[6]	0.7	59
Road ditches draining cropped land[6]		144
Forest[6]	0.2	78
Soybeans[7]	1.5	

[1]Timmons and Holt, 1977
[2]Young and Holt, 1977
[3]Burwell et at., 1975
[4]Crow et al., 1979
[5]Olness et al., 1975
[6]Young, R.A., Unpublished data, Morris, MN
[7]Logan and Stiefel, 1979
(after Young et al., 1982)

A COD to BOD conversion ratio of 4.57 $\pm$ 1.15 SD may be employed depending on the type of operation and the animal feeding ration for feedlots in northern states (Young et al., 1982).

Table 1.1.6. Background Runoff Concentrations For Total Nitrogen From Rural Land Uses.		
Source	Total Nitrogen Concentration (mg/l)	Reference
Native prairie–Minnesota	5	Timmons and Holt (1977)
Native prairie-Oklahoma	9	Crow et al. (1979)
Continuous corn-Minnesota	91	Young and Holt (1977)
Rotation corn-Minnesota	44	Young and Holt (1977)
Continuous corn-Minnesota	131	Burwell et al. (1975)
Wheat-Oklahoma	2	Olness et al. (1975)
Oats-Minnesota	49	Burwell et al. (1975)
Rotation pasture-Oklahoma	2	Olness et al. (1975)
Continuous pasture-Oklahoma	3	Olness et al. (1975)
Alfalfa-Oklahoma	4	Olness et al. (1975)
Alfalfa-Minnesota	60	Burwell et al. (1975)
Fallow-Minnesota	186	Burwell et al. (1975)
Aspen-birch forest-Minnesota	0	Timmons and Holt (1977)
(after Young et al., 1987)		

Reduction in animal waste pollutant concentrations in runoff moving as overland flow through a vegetative buffer is calculated below from regression models determined from field measurement studies (Young et al., 1982).

$$D_{c1} = -27.9 + 42.8 \log T_c \tag{1.1.28}$$

$$D_{p1} = -49.3 + 50.5 \log T_c \tag{1.1.29}$$

$$DN1 = -16.8 + 42.3 \log T_c \tag{1.1.30}$$

Where:

D_{c1} = percent reduction in COD concentration
(if < 0, $D_{c1} = 0$)

D_{p1} = percent reduction in TP concentration
(if < 0, $D_{p1} = 0$)

D_{n1} = percent reduction in TN concentration
(if < 0, $D_{n1} = 0$)

A similar relationship was developed for channelized flow for grassed waterways functioning a buffer areas (Young et al., 1982).

$$D_{c2} = 15.95 + 0.033\ T_c \qquad (1.1.31)$$

$$D_{p2} = 21.2 + 0.036\ T_c \qquad (1.1.32)$$

$$D_{n2} = 25.5 + 0.047\ T_c \qquad (1.1.33)$$

Where:

D_{c2} = percent reduction in COD concentration
(if 0, $D_{c2} = 0$)

D_{p2} = percent reduction in TP concentration
(if 0, $D_{p2} = 0$)

D_{n2} = percent reduction in TN concentration

TC = time of contact (sec) see equation 1.1.35

Where buffer areas are comprised of portions having overland flow through grassed waterways, the net reduction in animal waste pollutant concentration is calculated below:

$$C_R = C_F \times [1-(D_1/100)] \times [1-(D_2/100)] \qquad (1.1.34)$$

Where:

C_R = net reduced pollutant concentration (mg/1)
C_F = pollutant concentration at feedlot edge (mg/1)
D_1 = reduction in pollutant concentration in overland flow (percent)
D_2 = reduction in pollutant concentration in grassed waterway (percent)

If C_R equal or exceeds 100 percent, complete attenuation of pollutant has occurred, and the buffer strip is sufficient in removing any potential pollution hazard.

Attenuation of animal waste pollutant load from the feedlot edge to the watercourse is a function of infiltration, filtration, settling, and adsorption. Reduction of pollutant concentration depends on vegetative cover effects and time of contact (T_c).

$$T_c = L/V \tag{1.1.35}$$

Where:

T_c = time of contact (sec)
L = distance from feedlot edge to discharge point (ft)
V = flow velocity (ft/sec)

Where:

$\log V = 0.5 \ \log \ S\text{-}C$
S = slope (percent)
C = surface condition constant (Table 1.1.1)

An additional process for the reduction of feedlot pollution concentration may take place by dilution from the runoff of adjacent areas (Areas 3a-3f):

$$C_T V_T = C_R V_F + C_3 V_3 \tag{1.1.36}$$

Where:

C_T = final pollutant concentration at discharge point (mg/1)
V_T = total runoff volume at discharge point (ac-in)
C_R = reduced pollutant concentration after filtration (mg/1)
C_3 = pollutant concentration of runoff from Area 3 (mg/1)
V_3 = runoff volume from Area 3 (ac-in)

A more detailed, process-based model for erosion and sedimentation estimation is the Water Erosion Prediction Project (WEPP). A large team of researchers have been collaborating with field and watershed erosion experiments and model testing (Flanagan et al., 1995). Numerous papers on WEPP have been published in Transactions of the American Society of Agricultural Engineers.

1.1.3 NONPOINT SOURCES

Land use practices that result in nonpoint sources of pollution are difficult to manage. Land use changes may affect infiltration, runoff volumes, peak flows, and stream morphometry. Such changes are termed hydromodification. Habitat for fish and wildlife can be adversely affected by hydromodification.

Typical pollutants originating from nonpoint sources include sediment, nitrogen, phosphorus, fecal coliform bacteria, and pesticides.

An extensive discussion of the importance and magnitude of nonpoint sources is available in Novotny and Olem (1994). Annual June literature reviews in the journal Water Environment Federation on nonpoint source pollution control research have been compiled by Line et al.(1995, 1996, 1997).

1.1.3.1 Urban and Industrial Sites

Urban Pollutant Runoff. The findings of the Nationwide Urban Runoff Program (NURP) are particularly applicable to screening-level estimation of urban NPS pollutant loads. The NURP study (USEPA, 1983) included 28 projects with wide geographic representation in sites. Pollutant runoff concentrations and storm runoff volumes were monitored to assess urban land use pollutant export. Projects reported results in a consistent, transferable manner making statistical data analysis possible across projects and regions. The general results of the NURP study showed:

1. Urban pollutant runoff concentrations were essentially independent of hydrologic regions or geography.

2. Pollutant concentrations were essentially uncorrelated with storm runoff volumes especially for dissolved constituents and somewhat more correlated tot runoff volume for particulate constituents.

3. Pollutant loads were highly correlated with storm runoff volumes.

We provide an event-based urban pollutant loading function modified from Schuler (1987): The urban loading function is not restricted to dissolved constituents as the rural model given in equation 1.1.15. The urban model is, for source area k, at time t:

$$LD_{kt} = 0.1\ Cd_{kt}\ Q_{kt}\ A_k \tag{1.1.37}$$

Where:

LD = pollutant load (dissolved or solid phase) in kg
Cd = pollutant runoff concentration (mg/l)
Q = storm event source area runoff depth (cm)
A = source area (ha)

The pollutant concentration (Cd) may be determined from site-specific monitoring studies or the default values listed below in Table 1.1.7. These median and 90th percentile concentrations are taken from the range of EMC values pooled for all the NURP projects except for open/non-urban sites.

The median urban site values in Table 1.1.7 are recommended for pollutant runoff modeling. The 90th percentile values may serve as an upper conservative estimate for the EMC if desired. To the left of the EMCs are the ranges of coefficient of variation (CV = standard deviation divided by the mean) found for monitored runoff values. Higher CV values means lower confidence in the mean point estimate. These general values are given since land use category was found to explain little of the variability found in urban pollutant runoff.

Storm event runoff (Q) may be determined from the CNE (See section 1.1.1.3) or regression model by Schuler (1987). Table 1.1.8 provides a list of curve numbers for urban land use types. Each method considers the percentage of the source area covered by impervious surfaces such as paved roads, parking lots, or rooftops.

Table 1.1.7. Urban Event Mean Concentrations for Several Pollutants and their Variability

Constituent	Event to Event Variability In EMC's (Coef Var)	Site Median EMC	
		For Median Urban site	For 90th Percentile Urban Site
TSS (mg/1)	1-2	100	300
BOD (mg/l)	0.5-1.0	9	15
COD (mg/l)	0.5-1.0	65	140
Tot. P (mg/l)	0.5-1.0	0.33	0.70
Sol. P (mg/l)	0.5-1.0	0.12	0.21
TKN (mg/l)	0.5-1.0	1.50	3.3
NO2+3 -N (mg/l)	0.5-1.0	0.68	1.75
Tot. Cu (ug/l)	0.5-1.0	34	93
Tot. Pb (ug/l)	0.5-1.0	144	350
Tot. Zn (ug/l)	0.5-1.0	160	500
(after EPA, 1983)			

Table 1.1.8. Urban Runoff Curve Numbers					
Cover Description		Curve Numbers for Hydrologic Soil Group			
Cover Type and Hydrologic Condition	Average Percent Impervious	A	B	C	D
Fully Developed Urban (vegetation established					
Open space (lawns, parks, golf courses, cemeteries, etc.)[3]					
Poor condition (grass cover <50%)		68	79	86	89
Fair condition (grass cover 50% to 70%)		49	69	79	84
Good condition (grass cover >75%)		39	61	74	80
Impervious areas:					
Paved parking lots, roofs, driveways, etc.		98	98	98	98
Streets and roads:					
Paved; curbs and storm sewers (excluding right-of-way)		98	98	98	98
Paved: open ditches (including right-of-way)		83	89	92	93
Gravel (including right-of-way)		76	85	89	91
Dirt (including right-of-way)		72	82	87	89
Western desert urban areas					
Natural desert landscaping (pervious areas only)[4]		63	96	96	96
Artificial desert landscaping (impervious weed, barrier, desert shrub		96	96	96	96
Urban districts:					
Commercial and business	85	89	92	96	95
Industrial	72	81	88	91	93
Residential districts by average lot size:					
1/8 acre of less (townhouses)	65	77	85	90	92
¼ acre	38	61	75	83	87
1/3 acre	30	57	72	81	86
½ acre	25	54	70	80	85
1 acre	20	51	68	79	84
2 acres	12	46	65	77	82
Developing Urban Areas					
Newly graded areas (previous areas only, no vegetation)[5]		77	86	91	94
Idle lands (CN's are determined using cover types)					

[1]Average runoff conditions and $I_a = 0.2S$. [2]The average percent impervious area shown was used to develop the composite CN's. Other assumptions are as follows: impervious areas are directly connected to the drainage system, impervious areas have a CN of 98, and pervious areas are considered equivalent to open space in good hydrologic condition. CN's for other combinations of conditions may be computed for other combinations of conditions may be computed using Figure 2-3 or 2-4. [3]CN's shown are equivalent to those of pasture. Composite CN's may be computed for other combinations of open space cover type. [4]Composite CN's for natural desert landscaping should be computed using Figures 2-3 or 2-4 based on the impervious area percentage (CN = 98) and the pervious area CN. The pervious area CN's are assumed equivalent to desert shrub in poor hydrologic condition. [5]Composite CN's to use for the design of temporary measures during grading and construction should be computed using Figure 2-3 or 2-4, based on the degree of development (impervious area percentage) and the CN's for the newly graded pervious areas.
(After SCS, 1986)

Additional studies on urban runoff monitoring have been conducted (Marsalek and Schroter, 1984; Bannerman et al., 1983; Marsh 1993). Line et al. (1997) monitored storm runoff from 20 industrial sites including a chemical remanufacturer, furniture manufacturer, junkyard, landfill, metal fabricator, paint manufacturer, scrap and metal recycler, textile manufacturer, vehicle maintenance facility, and wood preserver. Characterization of these sites is given in Table 1.1.9. Table 1.1.10 shows the concentration of metals in industrial runoff from industrial sites and the concentration of conventional water quality parameters in industrial runoff is shown in Table 1.1.11.

1.1.3.2 Forestry

An undisturbed forested watershed delivers very small quantities of nutrients and little sediment to waterways.

Nutrient concentrations from the undisturbed forest are generally termed the uncontrollable background concentrations.

Deforestation and other vegetative removal has been shown to increase sediment and nutrient loading in a watershed. Phosphorus, in particular, is strongly associated with soil particulate matter. The removal of vegetative cover promotes erosion and increases both the sediment and phosphorus load in nearby aquatic systems. Terrestrial vegetation contains a large nutrient pool in some ecosystems. The destruction of this vegetation can release the stored nutrients, and make them available for leaching or runoff.

Forest practices such as roadbuilding and harvesting trees can result in erosion and sediment delivery to streams. Soil erosivity and slope are important factors that affect soil loss and sedimentation. Maintaining a streamside buffer zone is an important water quality practice. The buffer zone can reduce the transport of nutrients and sediment to streams and also provide habitat for fish and wildlife.

1.1.3.3 Agriculture

Agricultural practices can be important nonpoint sources of nutrients. Nitrogen and phosphorus fertilizers can be applied in large quantities to many croplands. Under poor management they can contribute to the nutrient load of the basin. Commercial feedlots and other intensive animal production facilities also produce wastes of high nutrient content, which often enter waterways via surface or subsurface flow.

Industrial Group	Site ID	Business Size*	Area of Site, ha	Drainage Area, ha	Impervious Area, %	Slope, %	Exposed Material	Sampling Data
Chemical Repackager	CR I	Medium	6.5	6.5	98	1-2	No	12/14/93
Furniture	FM I	Medium	4.9	1.6	25	2-4	No	12/14/93
Furniture	FM II	Large	6.9	1.2	100	0-1	No	5/15/93
Junkyard	JY I	Medium	4.0	0.8	25	4-6	Yes	11/05/93
Junkyard	JY II	Small	4.0	1.6	20	3-5	Yes	2/24/94
Landfill	LF I	Medium	40.0	8.1	0	3-25	Yes	1/17/94
Landfill	LF II	Large	120.0	2.4	0	4-6	Yes	2/24/94
Metal Fabricator	MF I	Small	2.4	1.6	50	1-3	Yes	1/12/94
Metal Fabricator	MF II	Medium	0.8	0.4	95	0-1	No	3/24/94
Paint Manufacturer	PM I	Small	1.2	1.2	25	3-5	No	12/04/93
Paint Manufacturer	PM II	Medium	1.6	1.2	70	0-1	No	11/05/93
Scrap and Recycler	SR I	Medium	4.5	2.4	10	1-3	Yes	2/10/94
Scrap and Recycler	SR II	Medium	4.0	1.6	50	2-5	Yes	1/12/94
Scrap and Recycler	SR III	Small	2.4	1.2	100	0-1	Yes	4/27/94
Textile Manufacturer	TM I	Large	8.1	0.8	100	0-1	No	12/29/93
Textile Manufacturer	TM II	Medium	2.4	0.4	100	0-1	No	3/01/94
Vehicle Maintenance	VM I	Medium	2.4	1.6	20	2-3	Yes	11/05/93
Vehicle Maintenance	VM II	Large	2.4	2.4	100	0-1	No	9/16/93
Wood Preserver	WP I	Medium	2.0	0.4	10	0-1	Yes	11/27/93
Wood Preserver	WP II	Medium	4.9	3.2	30	1-2	Yes	5/03/94

Table 1.1.9 Characterization of the 20 Study Sites

*Cursory estimate of how facility compares in production to other North Carolina businesses in the same industrial group

Table 1.1.10 Concentrations of Metals and Other Analytes in First Flush Runoff Sample and Storm Rainfall[a]										
Method[c] Site	206.2 As	213.2 CD	218.2 Cr	220.2 Cu	239.1 Pb	245.1 Hg	249.2 Ni	289.1 Zn	Other Compounds[b]	Rainfall
CR I	<10	<2	44	34	22	<0.2	38	220	ND	14.5
FM I	<10	5	12	25	20	<0.2	6	220	*m*	14.0
FM II	<10	<2	67	29	12	<0.2	11	473	ND	8.1
JY I	<10	<2	25	27	67	<0.2	<10	398	*m*	10.2
JY II	<10	4	23	97	330	0.4	34	678	ND	40.6
LF I	<10	<2	7	45	25	<0.2	<10	84	ND	6.4
LF II	<10	<2	12	16	12	0.2	20	792	*a, b, p*	35.6
MF I	<10	<2	73	57	100	<0.2	49	1051	ND	16.3
MF II	<10	6	15	29	41	<0.2	17	805	ND	19.8
PM I	<10	<2	10	11	7	<0.2	<10	60	ND	33.0
PM II	<10	<2	<5	5	<5	<0.2	<10	154	ND	22.9
SR I	<10	<2	<5	110	37	<0.2	28	190	*t2, al*	10.2
SR II	<10	10	170	530	660	3	78	2689	*an, m*	16.3
SR III	<10	5	28	99	59	<0.2	28	1797	*a*	13.7
TM I	<10	<2	<5	6	<5	0.3	<10	120	ND	7.6
TM II	<10	6	<5	7	24	<0.2	12	895	*t, t1, t2*	33.0
VM I	<10	<2	<5	5	<5	<0.2	<10	154	ND	10.2
VM II	<10	<2	16	120	34	<0.2	<10	219	*e*	8.1
WP I	330	<2	610	280	48	<0.2	<10	260	ND	33.0
WP II	140	<2	1700	780	150	<0.2	200	592	ND	33.0
Mean[d]	24 ± 77	2 ± 3	141 ± 382	116 ± 194	82 ± 151	0.2 ± 0.7	26 ± 45	593 ± 638		19.3 ±
Standards[e]	50	2	50	7 (A)	25	0.012	88	50 (A)		
NURP				34	144			160		

[a] Metal concentrations in ppb; rainfall in mm; "<" indicates concentration below specified detection limit; ND indicates none measured above detection limits
[b] Other compounds are: *a*, acetone; *an*, acrolein; *al*, aldrin; *b*, benzoic acid; *e*, endrin; *m*, methylene chloride; *p*, phenol; *t*, tetrachloroethylene; *t1*, 1,1,1-trichloroethane; *t2*, trichloroethylene
[c] US EPA, 1983a
[d] Means ± SD
[e] North Carolina standards or action levels (A) for all fresh water
[f] Median event mean concentrations from the Nationwide Urban Runoff Program (US EPA 1983b)

Table 1.1.11 Concentrations (mg/l) of Conventional Water Quality Parameters in First Flush Runoff Samples										
				Nitrogen Forms			Phosphorus		Solids	
Method[a] Site	5210B BOS_5	410.4 COD	5520B Oil & Grease	4500 NH_3	353.1 NO_2 + NO_3	351.2 TKN	365.4 Total	365.4 Dissolved	2540C Dissolved	2540D Suspended
CR I	NA	76.0	<5	<0.04	1.29	2.0	0.55	0.31	170	362
FM I	2.7	5.6	<5	0.04	0.37	0.4	0.24	0.14	50	105
FM II	14.3	130.0	<5	0.15	0.52	1.7	0.18	0.14	102	76
JY I	NA	56.0	6	0.08	3.23	1.4	0.49	0.28	108	198
JY II	5.5	41.4	<5	0.08	3.15	1.2	0.26	<0.05	208	2770
LF I	13.0	23.9	<5	0.72	0.94	1.6	0.32	0.09	85	310
LF II	520.0	870.0	<5	6.40	0.44	7.0	0.87	0.31	1570	228
MF I	9.9	22.2	<5	0.23	0.65	1.0	0.70	0.24	166	668
MF II	52.6	260.0	<5	0.89	2.28	5.8	0.69	0.62	166	128
PM I	4.9	35.3	<5	<0.04	0.22	0.8	0.35	0.13	121	143
PM II	NA	28.9	<5	0.12	0.42	0.5	0.39	0.27	48	24
SR I	28.9	230.0	31	0.50	0.20	3.8	0.18	<0.05	434	88
SR II	43.5	510.0	<5	0.05	0.40	2.6	1.31	0.61	454	627
SR III	130.0	530.0	28	2.30	0.13	15.6	2.88	1.01	322	402
TM I	NA	20.4	<5	0.20	0.77	0.7	0.22	0.12	46	<1
TM II	NA	11.3	<5	0.04	0.30	0.4	0.29	0.06	72	6
VM I	NA	64.0	6	<0.04	0.58	1.4	0.40	0.33	65	38
VM II	28.9	130.0	51	0.04	0.56	1.1	0.41	0.28	36	93
WP I	NA	42.9	<5	0.04	0.55	1.8	1.06	0.55	138	912
WP II	14.6	42.9	<5	<0.04	0.29	3.5	4.21	2.17	150	3260
Mean[b]		156.6 ± 221.8	6	0.6	0.9	2.7	0.8	0.4	225.6	521.9
			± 14	± 1.4	± 0.9	± 3.4	± 1.0	± 0.5	± 329.8	± 868.2
NC Problem				0.05	0.30	0.5	0.05			
NURP EMC[f]		65			0.68	1.5	0.33			100

[a] US EPA, 1983a and APHA et al., 1989
[b] Means ± SD
[c] Median event mean concentrations from the Nationwide Urban Runoff Program (US EPA 1983b)

Table 1.1.12 Export Coefficients Compiled from Literature Reviews by Reckhow and Dodd			
Crop	TN	TP	Studies
	kg/ha-yr	kg/ha-yr	n
Hay	4.09	0.64	1
Alfalfa	5.6	0.91	5
Cotton	10.2	4.45	2
Tobacco	13.3	4.5	3 and 7
Corn	11.32	2.74	15
Soybeans	26	13	3

1.1.3.4 Construction

Erosion, sedimentation, and hydromodification from construction sites can be an important cause of water pollution and loss of designated uses in a waterway. Residential construction without erosion control BMPs can cause a streambed to be filled with sediment. Highway construction without effective BMPs can fill small lakes with sediment, making them useless for their intended uses.

1.1.4 POINT SOURCES

1.1.4.1 Municipal and Industrial Waste Water

Point sources are, in principle, relatively straightforward to control. Wastewaters originate from households, industries, and combined sewers. They are collected and transported to the treatment plant where they are processed. The effluent then flows to streams where it is diluted by streams, lakes, or estuaries. Occasionally part of the water supply volume for one community is the discharge from the wastewater treatment plant from the community upstream. Additional methods of disposal include irrigation, infiltration, evaporation from lagoons, and submarine outfalls extending into the ocean (Viessman and Hammer, 1985).

Wastewater volume and concentration information by treatment process provides information for calculating wastewater pollutant loading (Table 1.1.13).

1.1.5 NONPOINT SOURCE SIMULATION MODELS

1.1.5.1 Estimating On-Site and Off-Site Impacts

On-site benefits are highly desirable, yet unless the needed off-site benefits are derived from the collective implementation of management measures and practices across the watershed,

Table 1.1.13 Total Nitrogen and Total Phosphorus Concentration in Wastewater Treatment Plant Effluent

Treatment Type	Mean Nitrogen (mg/l)	Mean Phosphorus (mg/l)
Activated Sludge	15.8	5.91
Tricking Filter	17.9	7.25
Phosphorus Removal	4.0	2.85
Primary Settling and Digestion	23.8	8.71
Oxidation Pond	17.1	6.42
Sand Filter	--	5.02

(after Gakstatter et al. 1978 and Reckhow and Chapra 1983)

then implementation has not been fully successful. It is important to estimate the collective impacts of all management activities in the watershed to gage whether water quality goals will be achieved. In watersheds with easily characterized problems (e.g., bacterial contamination is due to a few obviously polluting animal operations in a watershed that has no other identifiable sources of pathogens) it may be very easy to project that water quality benefits will be achieved through implementation of the management measures for nutrient management, erosion and sediment control, and facility wastewater and runoff, for example. However, in a watershed with multiple land uses where agriculture is considered to contribute about one-third or so of the pollutants, it is more complicated to estimate the combined impacts of a variety of management measures and practices on a fairly large number of diverse farming operations. In this type of situation, computer modeling may be needed.

A variety of models exist to help assess the benefits of implementing practices at the farm level, some of which could also be used on small watersheds.
These include the following:

GLEMS (Knisel et al., 1991) simulates the effects of management practices and irrigation options on edge of field surface runoff, sediment, and dissolved and sediment attached nitrogen, phosphorus, and pesticides. The model considers the effects of crop planting date, irrigation, drainage, crop rotation, tillage, residue, commercial nitrogen and phosphorus applications, animal waste applications, and pesticides on pollutant movement. The model has been used to predict the movement of pesticides (Zacharias et al., 1992) and nutrients and sediment from various combinations of land uses and management (Knisel and Leonard, 1989, Smith et al., 1991).

EPIC (Sharpley and Williams, 1990) simulates the effect of management strategies on edge of field water quality and nitrate nitrogen and pesticide leaching to the bottom of the soil profile. The model considers the effect of crop type, planting date, irrigation, drainage, rotations, tillage, residue, commercial fertilizer, animal waste, and pesticides on surface and shallow ground water quality. The EPIC model has been used to evaluate various cropland management practices (Sugiharto et al., 1994; Edwards et al., 1994).

NLEAP (Follet et al., 1991) evaluates the potential of nitrate nitrogen leaching due to land use and management practices. The NLEAP model has been used to predict the potential for nitrogen leaching under various management scenarios (Wylie et al., 1994; Wylie et al., 1995).

PRZM (Mullens et al. 1993) simulates the movement of pesticides in unsaturated soils within and immediately below the root zone. Several different field crops can be simulated an up to three pesticides are modeled simultaneously as separate parent compounds or metabolites. The PRZM model has been used under various conditions to assess pesticide leaching under fields (Zacharias et al., 1992; Smith et al., 1991).

WEPP (Flanagan and Nearing, 1995) simulates water runoff, erosion, and sediment delivery from fields or small watersheds. Management practices including crop rotation, planting and harvest date, tillage, compaction, strip cropping, row arrangement, terraces, field borders, and windbreaks can be simulated. The WEPP model has been applied to various land use and management conditions (Tiscareno-Lopez et al., 1993; Liu et al., 1997)

DRAINMOD (Skaggs, 1980) simulates the hydrology of poorly drained, high water table soils. Breve (1994) developed DRAINMOD-N, a nitrogen version of the model to evaluate nitrogen dynamics in artificially drained soils. The DRAINMOD model has been used to predict pollutant losses associated with various drainage management scenarios (Deal et al., 1986).

BARNY (Vermont NRCS, 1985) is a spreadsheet model that estimates total phosphorus losses from dairy barnyards and estimates phosphorus loads entering waterways. Other similar models include STACK, PHRED, and MILKHOUSE. These models have had limited use.

SWRRBWQ (Arnold et al., 1990) simulates the effect of agricultural management practices such as crop rotation, conservation tillage, residue, nutrient, and pesticide management; and improved animal waste application methods on water quality. The SWRRB model has been used on several watersheds to assess management practices and to test its validity (Arnold and Williams, 1987; Bingner et al., 1987).

AGNPS (Young et al. 1994) is a spatially-distributed model for estimating pollutant runoff from agricultural watersheds. Within cells, the model can evaluate practices such as feedlot management, terraces, vegetative buffers, grassed waterways, and farm ponds. Simulated nutrient, sediment, and pesticide concentrations and yields are available for any cell within the watershed. The AGNPS model has been applied to many field and watershed size areas to estimate pollutant runoff from various land uses and management practices (Line et al., 1997: Sugiharto et al., 1994; Bingner et al., 1987)

ANSWERS (Beasley, 1980) is a spatially-distributed watershed model. The model is primarily a runoff and sediment model as soil nutrient processes are not simulated. The ANSWERS model has been applied to several small fieldsized areas with various management practices (Griffin et al., 1988; Bingner et al., 1987).

NTRM (Shaffer and Larson, 1985) simulates the impact of soil erosion on the short and long-term productivity of soil, and is intended to assist with evaluation of existing and proposed soil management practices in the subject areas of erosion, soil fertility, tillage, crop residues, and irrigation. The NTRM model has been applied to evaluate effects of conservation tillage, supplemental nitrogen and irrigation practices (Shaffer, 1985) and moldboard plow and chisel plow tillage (Shaffer et al., 1986) on soil erosion and productivity. This model has had limited use.

A series of protocols has been developed by EPA to assist in the development of TMDLs and implementation plans to achieve the TMDLs (EPA, 1997 - **DRAFT at this point)**. These protocols focus primarily on the application of computer models that simulate watershed conditions and the changes that could result from implementation of various land management scenarios. Most models contain default values for the quantity of pollutants that are delivered in runoff.

1.1.6 REFERENCES

Arnold, J. G. and R. R. Williams. 1987. "Validation of SWRRB - Simulation for Water Resources in Rural Basins. *J. Water Resources Planning and Management* 13:243-256.

Arnold, J. G., J. R. Williams, A. d. Nicks, and N. B. Sammons. 1990. SWRRB: A Basin Scale Simulation Model for Soil and Water Resources management. Texas A&M University Press, College Station, Texas.

Bannerman et al. 1983. "Sources of Pollutants in Wisconsin Stormwater," *Water Sci. Technol*. 28:713.

Beasley, D. B., L. F. Huggins, and E. J. Monke. 1980. "ANSWERS: A Model for Watershed Planning," *Trans. of the ASAE* 23:938-1442.

Binger, R. L., C. E. Murphree, and C. K. Mutchler. 1987. *Comparison of Sediment Yield models on Various Watersheds in Mississippi.* ASAE Paper 87-2008, American Society of Agricultural Engineers, St. Joseph, MI.

Breve, M. A., R. W. Skaggs, J. E. Parsons, and J. W. Gilliam. 1997. "DRAINMOD-N, A Nitrog Model for Artificially Drained Soils," *Trans of the ASAE*, Vol 40(4): 1067-1075.

Chow, V.T. 1964. *Handbook of Hydrology*. McGraw Hill, New York, NY.

Davis, C. V. and K.E. Sorensen. 1969. *Handbook of Applied Hydraulics*. McGraw-Hill, New York, NY.

Deal, J.C., J.W. Gilliam, R.W. Skaggs, and K.D. Konyha. 1986. "Prediction of Nitrogen and Phosphorus Losses as Related to Agricultural Drainage System Design," *Agriculture Ecosystem and Environment* 18:37-51.

Delwiche, L.L.D., and D.A Haith, 1983. "Loading Functions for Predicting Nutrient Losses from Complex Watersheds," *Water Resources Bulletin* 19 (6): 951-959.

Dillon, P.J. and F.H. Rigler. 1975. "A Simple Method for Predicting the Capacity of a Lake for Development Based on Lake Trophic Status," *J. Fish. Res. Bd. Can.* 32:1519-1531.

Dunne, T. and L.B. Leopold. 1978. *Water in Environmental Planning*. W.H. Freeman and Co. San Francisco, 818p.

Edwards, D.R., V.W. Benson, J.R. Williams, T.C. Daniel, J. Lemunyon, and R.G. Gilbert. 1994. "Use of the EPIC Model to Predict runoff Transport of Surface-Applied Inorganic Fertilizer and Poultry Manure Constituents," *Trans. of the ASAE* 37(2):403-409

Flanagan, D.C. and M.A. Nearing. Eds. 1995. *USDA-Water Erosion Prediction Project: Hillslope Profile and Watershed Model Documentation*. NSERL Report No. 10. West Lafayette, IN.

Follet , R.F., D.R. Kenney, and R.M. Cruse. 1991. *NLEP or Nitrogen Leaching and Economic Analysis Package*. Soil Science of America, Inc., Madison, WI.

Green, W. H. and G. A. Ampt. 1911. "Studies in Soil Physics I. The Flow of Air and Water Through Soils," *J. of Agri. Sci.* 4:1-24.

Haan, C.T. 1972. “A Water Yield Model for Small Watersheds,” *Water Resources Research*. 8(1):58-69.

Haith, D.A. 1985. “An Event-Based Procedure for Estimating Monthly Sediment Yields,” *Transactions of the ASAE* 28(6):1916-1920.

Haith, D.A. and L.L. Shoemaker. 1987. “Generalized Watershed Loading Functions for Stream Flow Nutrients,” *Water Resources Bulletin*. 12(2):471-478.

Haith, D.A. 1985. “An Event-Based Procedure for Estimating Monthly Sediment Yields,” *Transactions of the ASAE* 28(6): 1916-1920.

Haith, D.A. and L.L. Shoemaker, R.L. Doneker, and L.L.D. Delwiche. 1984. Agricultural Nonpoint Source Control of Phosphorus and Sediment--A Watershed Evaluation, Vol. IV. US EPA, Ada, Oklahoma.

Haith, D.A. and L.J. Tubbs. 1981. “Watershed Loading Functions for Nonpoint Sources,” *J. Env. Eng. Div. ASCE* 107(EE1):121-137.

Hamon, W.R. 1961. “Estimating Potential Evapotranspiration,” *Proceedings of the American Society of Civil Engineers, Journal of Hydraulics Division* 87(HY3):107-120.

Hendrick, R.L., B.D. Filgate, and W.A. Adams. 1971. “Application of Environmental Analysis to Watershed Snowmelt,” *J. of Applied Meterology*. 10:418-429.

Hershfield, D.M. 1961. Rainfall Frequency Atlas of the United States; U.S. Weather Bureau Technical Paper 40, Washington, DC.

Knisel, W.G., R.A. Leonard, and F.M. Davis. 1991. “Water Balance Components in the Georgia Coastal Plain: A GLEAMS Model Validation and Simulation. *J. of Soil and Water Conservation* 46(6):450-456.

Line, D.E., J. Wu, J. A. Arnold, G. D. Jennings, and A.R. Rubin. 1997. “Water Quality of First Flush Runoff from 20 Industrial Sites,” *Water Environ. Res*. 69:305.

Liu, B.Y., M.A. Nearing, C. Baffault, and J.C. Ascough. 1997. “The WEPP Watershed Model: Three Comparisons to Measured Data from Small Watersheds,” *Trans. Of the ASAE* 49:945-952.

McCuen , R. H. 1982. *A Guide to Hydrologic Analysis Using SCS Methods*. Prentice-Hall, Englewood Cliffs, NJ.

Marsh, J. M. 1993. "Assessment of Nonpoint Source Pollution in Stormwater runoff from Louisville, Jefferson County Kentucky, USA. *Environ. Contam. Toxicol*. 25:446.

Manny, B. A., R. G. Wetzel and W. C. Johnson. 1975. "Annual Contribution of Carbon, Nitrogen, Phosphorus to a Hard-Water Lake b Migrant Canada Geese," *Verh. Int. Limnol*. 19:949-951.

Mills, W.B., D.B. Porcella, M.J. Ungs, S.A. Cherini, K.V. Smmers, L. Mok, G.L. Rupp, G.L. Bowie, and D.A. Haith. 1985. "Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water," EPA-600/6-85-002, US EPA Washington, DC.

Mockus, V. 1972. Estimation of Direct Runoff from Storm Rainfall, in: National Engineering Handbook, Section 4: Hydrology, US Soil Conservation Service, Washington, DC.

Monteith, J.L. 1977. "Climate and the Efficiency of Crop Production in Britain," *Phil. Trans. Res. Soc*. London b. 281:277-329.

Mullens, J. A. R. F. Carsel, J. E. Scarbrough, and I. A. Ivery. 1993. *PRZM-2, A Model forPredicting Pesticide Fate in the Crop root and Unsaturated Soil Zones: Users Manual for Release 2.0*. Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Athens, GA EPA/600/R-93/046.

New Jersey State Soil Conservation Committee. 1982. Standards for Soil Erosion and Sediment Control in New Jersey, NJ.

Nicks, A.D. 1974. "Stochastic Generation of the Occurance, Pattern and Location of Maximum Amount of Daily Rainfall," pp 154-171 in *Proc. Symp. On Statistical Hydrology,* Aug-Sept 1971, Tuscon, Ariz, Misc. Publicatoin No. 1275.

Novotny, V, and H. Olem. 1994. *Water Quality Prevention, Identification, and Management of Diffuse Pollution*, Van Nostram Reinhold, 1054 pp.

Ogrosky, H.O. and V. Mockus. 1964. Hydrology of Agricultural Lands in: *Handbook of Applied Hydrology*, V.T. Chow, ed, McGraw-Hill Publishing Co., New York, NY, Chapter 21.

Priestley, C.H.B. and R.J. Taylor. 1972. "On the Assessment of Surface Heat Flux and Evapotranspiration Using Large Scale Parameters," *Monthly Weather Review*. 100:81-92.

Reckhow, K.H., and S.C. Chapra. 1983. *Engineering Approaches for Lake Management, Vol I: Data Analysis and Empirical Modeling*. Butterworth, Boston. 340 pp.

Reckhow, K.H., J.P. Hartigan and S. Coffey. 1988. "Lake Nutrient Budget Development for State-Level Applications," *Proceedings of a National Conference on Enhancing States' Lake Management Programs*, May 12-13, Chicago, IL. Northeast Illinois Planning Commission, 400 W. Madison Avenue, Suite 200 Chicago, IL 60606.

Richardson, C.W., G.R. Foster, and D.A. Wright. 1983. "Estimation of Erosion Index from Daily Rainfall Amount," *Trans. Of ASAE* 26(1):153-156.

Ritchie, J.T. 1972. "A Model for Predicting Evapotranspiration from a Row Crop with Incomplete Cover," *Water Resources Research* 8(5):1204-1213.

Schueler, T.R. 1987. Controlling Urban Runoff: A Practical Manual for Planning and Designing Urban BMPs. Metropolitan Washington Council of Governments.

Schwab, G.O., D.D. Fangmeier, W.J. Elliot, and R.K. Frevert. 1993. *Soil and Water Conservation Engineering*. John Wiley and Sons, New York. 507p.

Shaffer, M.J. and W.E. Larson, eds. 1985. NTRM: *A Soil-Crop Simulation Model for Nitrogen, Tillage, and Crop Residue Management*. USDA Agricultural Research Service, St. Paul, MN.

Shaffer, M.J., S.C. Gupta, D.R. Linden, J.A.E. Molina, C.E. Clapp, and W.E. Larson. 1986. Simulation of Nitrogen, Tillage, and Residue Management Effects on Soil Fertility. In: *Analysis of Ecological Systems: State-of-the-art in Ecological Modelling*. W.K. Lauenroth. G.V. Skogerboe, and M. Flug, eds. Developments in Environmental Modelling 5. Elsevier Scientific Publishing Company.

Sharpley, A.N. and J.R. Williams, eds. 1990. *EPIC-Erosion/Productivity Impact Calculator: 1. Model Documentation*. U.S. Department of Agriculture Bulletin No. 1768. 234 pgs.

Skaggs, R.W. 1980. *A Water Management Model for Artificially Drained Soils*. North Carolina Agricultural Research Service Technical Bulletin 267, North Carolina State University, Raleigh, NC.

Smith, M.C., A.B. Bottcher, K.L. Campbell, and D.L. Thomas. 1991. Field Testing and Comparison of the PRZM and GLEAMS Models. *Trans. Of the ASAE* 34(3):838-847.

Soil Conservation Service. 1986. *Urban Hydrology for Small Watersheds Manual*. Washington DC, Technical Release 55. Second Edition. PB 87-101580/AS A08.

Soil Conservation Service. 1986. *Urban Hydrology for Small Watersheds*. (Computer Program on Diskette) Washington, DC, Technical Release 55, Second Ed. PB 87-101598/AS T99.

Stewart, B.A., D.A. Woolhiser, W.H. Wischmeier, J.H. Caro and M.N. Frere. 1975. Control of Water Pollution from Croplands, EPA-600/2-75-026., US EPA, Washington, DC.

Sugiharto, T., T.H. McIntosh, R.C. Uhrig, and J.J. Lavdinois. 1994. Modeling Alternatives to Reduce Dairy Farm and Watershed Nonpoint Source Pollution. *J. Environmental Quality* 23:18-24.

Sweeten, J.M. and D.L. Reddell. 1976. Nonpoint Sources. State-of-the-Art Overview. *American Society of Agricultural Engineers*, St. Joseph, MI. Paper No. 76-2653.

Tiscareno-Lopez, M., V. L. Lopes, J. J. Stone, and L. J. Lane. 1993. Sensitivity Analysis of the WEPP Watershed model for Rangeland Applications: I. Hillslope Processes. *Trans. Of the ASAE* 36:1659-1672.

Tung, Y.K. 1987. Uncertainty Analysis of National Weather Service Rainfall Frequency Atlas. J. of Hydraulic Eng. ASCE. 113(2): 179-189.

US Environmental Protection Agency. 1983. Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses.

Vermont NRCS. 1985. *BARNY, A dairy Barnyard Evaluation Tool. USDA-Natural Resources Conservation Service*. Winooski, VT.

Viessman W. and M. J. Hammer. 1985. *Water Supply and Pollution Control*, Harper Collins, New York, 797pp.

Williams, J.R., C.A. Jones, and P.T. Dyke. 1984. “A Modeling Approach to Determining the Relationship Between Erosion and Soil Productivity,” *Transactions ASAE* 27:129-144.

Wischmeyer, W.H. And D.D. Smith. 1978. “Predicting Rainfall Erosion Losses - A Guide to Conservation Planning,” USDA Handbook 537. GPO, Washington, DC.

Wylie, B.K., M.J. Shaffer, M.K. Brodohl. D. Dubois, and D.G. Wagner. 1994. “Predicting Spatial Distributions of Nitrate Leaching in Northeastern Colorado,” *J. Soil and Water Conservation* 49:288-293.

Wylie, B.K., M.J. Shaffer, and M.D. Hall. 1995. “Regional Assessment of NLEAP No_3-N Leaching Indices. *Water Resources Bulletin* 31:399-408.

Young, R.A., C.A. Onstad, D.D. Bosch, and W.P. Anderson. 1994. *Agricultural Nonpoint Source Pollution Model, Version 4.03 AGNPS User’s Guide*. USDA Agricultural Research Service, Morris, MN.

Young, R.A., C.D. Onstad, D.D. Bosch and W.P. Anderson. 1987. "AGNPS, Agricultural Non-Point-Source Pollution Model. A Watershed Analysis Tool," US Dept of Agriculture, Conservation Research Report 35. 80 p.

Young, R.A., M.A. Otterby, and A. Roos. 1982. An Evaluation System to Rate Feedlot Pollution Potential. Agricultural Research Service, US Dept. Of Agriculture, ARM-NC-17, Peoria, IL.

Zacharias, S., C. Heatwole, T. Dillaha, and S. Mostighimi. 1992. *Evaluation of GLEAMS and PRZM for Predicting Pesticide Leaching Under Field Conditions*. ASAE Paper 92-2541. American Society of Agricultural Engineers, St. Joseph, MI.

1.2 PRACTICAL LIMNOLOGY

1.2.1 INTRODUCTION

Limnology is often described as inland or freshwater oceanography. The term 'limnology' is derived from the Greek word *limnos*, which refers to pools, lakes, and swamps. Limnology is a field that fits the ecosystem perspective well because in order to understand or explain aquatic life and their interactions, a thorough knowledge of their physical and chemical environments is necessary. And the collective system of interactions between aquatic organisms and the chemical and physical processes of their environment is the aquatic ecosystem.

In limnology, as in other fields, physical processes are comparatively well understood and predictable compared with chemical and biological processes. Biological processes are the least predictable and where chemical processes interact with biological processes, predictions of chemical characteristics are also difficult. It follows, then, that while specific accurate predictions can be made using complex models of lake physical processes, only the most general informed guesses can often be made about the biota.

Environmental systems or ecological systems are more difficult to predict than social or economic systems. We have so much less direct knowledge of these ecological systems that consequences of our decisions about ecosystems are much less likely to be predicted accurately than decisions about our personal lives. This is due to two reasons, a lack of knowledge of many of the processes which influence ecosystems and a lack of empirical data about real systems. This workshop is designed to aid field personnel with the collection of useful data about real systems.

1.2.2 WHAT IS A SYSTEM AND WHAT GOOD IS IT?

For this discussion, any reference to a system will imply an **environmental** or **ecological system**. And these two terms will be used interchangeably. Furthermore, an **ecological system** will be considered synonymous with **ecosystem**. And an **ecosystem** will be considered in the sense of its original definition by Tansley in 1935 which is any system composed of interacting physical, chemical, and biological components, regardless of size.

A moment of consideration of that definition will show it to be vague and, short of applying the concept, it is trivial. However, Tansley had good reasons for formalizing the concept, although other scientists had implied the concept long before. Today, the ecosystem concept or perspective has achieved widespread acceptance as the only realistic approach to understanding ecological interactions. Sadly, it is rarely applied to practical problems in resource management.

1.2.3 WHAT ARE THE KINDS OF SYSTEMS?

Any dynamic system is composed of interacting elements or components. All of the ecological systems that occupy our interest are dynamic systems. Lakes and ponds are no exceptions. They can be viewed in a variety of ways. In one example, ecosystems are thermodynamic systems. **Open systems** exchange both energy and materials across their boundaries. All ecological systems (except for the microcosm) are open systems. The accounting necessary for application of thermodynamic principles to ecosystems is a complex but lively field of inquiry.

In another example, ecosystems are information systems where the currency of interest is not material or energy but information. The information is observed and described (modeled) using sophisticated mathematics. Ecological diversity is a simple example of such an approach applied to community structure.

At their simplest, ecosystems can be considered machines in which materials are placed into motion via energy flow through the system. The form of the materials is determined by the physical structures and organisms in the system. **Ponds and lakes, too, can be viewed as mechanisms in which sunlight enters the water causing thermal changes and activating plants which absorb chemicals during growth. Their later consumption by animals or decomposition releases energy and materials.** This apparent cycle, simply viewed here, is the centerpiece of nutrient cycling a recent topic of hot limnological discussion. But the components of this system are always the same, energy transfer allowing movement and conversion of materials. Complex water quality models such as CEQUAL employ this approach.

Again, energy inputs can be shown, for example, to drive the conversion of water from liquid to vapor which is transported to other locations in the system where the water is transformed to water again through another energy exchange. This hydrologic cycle is central to all aquatic systems and often determines the physical and energetic limits to the system in question. Other, different kinds of energy exchanges will enable inorganic materials to become transformed into organic structures. Then the decomposition of those structures will release some energy and enable the incorporation of those materials into other organic forms or into inorganic form. These systems have great complexity and yet still perform the same basic functions as the most simple conceptual system. However, those complex systems are much closer to reality and in order to use them, we must contend with the complexity.

Most often limnologists consider the system boundary of the reservoir system as the shoreline. But to paraphrase a comment by Barbara Speziale, every time one of us waters a lawn or flushes a toilet, the system boundary is redefined.

1.2.4 INTRODUCTION TO RESERVOIRS

1.2.4.1 History of Reservoirs

The first archeological evidence of dams (and reservoirs) is dated to approximately 3000 BC. These structures were constructed in the region recognized today as Jordan. A similar body of evidence and history of earthen and stone dams and reservoirs is found for other regions such as Europe, other parts of the Middle East, Southeast Asia, China, and Central America. These early reservoirs were constructed for the purposes of irrigation and water supply. This technology was a logical development where human needs for water surpassed naturally occurring supplies.

Hydropower from dams dates to antiquity. Watermills using flowing water were used to turn grinding stones in the first century BC. This technology was used widely and water power was crucial to industry early in the industrial revolution. Later, electricity generated from hydropower was able to export the energy of water motion to locations greatly removed from the stream.

As development continued and as population densities increased, the impact of floods on local and regional economies was greater and greater. Dams served dual purposes of providing protection from flooding as well as electrical power. Regional development was stimulated through large dam construction programs such as administered by the Tennessee Valley Authority.

In areas where farming was limited by water availability, dams and reservoirs provided water for irrigation. This was more prevalent in the western states of the U.S.A. The first very large dam project was Hoover Dam which was begun in 1931 and , when completed, was the largest dam in the world. Today, much larger dams have been completed and are planned all around the world.

The reasons for building reservoirs are several but not numerous. The reasons usually include flood protection, hydropower, water supply, irrigation, recreation, and wildlife habitat. Often in this country, a reservoir authorized for one set of purposes may eventually offer other economic opportunities as well. Most recently, because dam and lake construction is costly and controversial, dams have been authorized for multiple and sometimes conflicting purposes.

The economic benefit of reservoirs is also not constant. Local and regional development around and near reservoirs can greatly increase the economic interest for recreation. If sufficient change occurs, the authorized purpose may be modified. In 1997, Glen Canyon dam began experimental non-power releases to simulate the flooding that occurred seasonally in the river prior to its construction. If the benefits to habitat and recreation are proven sufficient, this operation may become normal. Elsewhere, a dam may be removed if sufficient economic purpose is discovered. These decisions are all important to aquatic habitats and water quality because they determine operational schedules and patterns of water movement.

1.2.5 TERMINOLOGY

1.2.5.1 Lakes and Streams

As may already have been discussed, lakes differ fundamentally from streams in that streams contain flowing water and are described as **lotic** and lake waters are standing and are described as **lentic**. A further distinction is made between saline or salt-water bodies **(marine)** and freshwater bodies **(lacustrine)**. The distinctions do not end, however, with these. Within a single lake, there are a variety of habitats and life zones within which biota reside for some or all of their lives. These regions often coincide, for good reasons, with physical or chemical limnological zones or regions of the lake, betraying the fundamental interdependence of aquatic life with their physical and chemical environments as well as among themselves.

1.2.5.2 Morphometry

Although the lakes managed by the Corps of Engineers are reservoirs, the comparison with other lakes is important because original limnological principles were developed in natural lakes and these do not always apply to reservoirs.

Lakes are shaped differently. Most people are familiar with the Great Lakes and their shape. And there are smaller northern lakes that are almost round in shape. Lakes shaped like those also occur in the coastal plain and in Florida. Many lakes have unusual characteristics because they were constructed as an engineering project. These **reservoirs** (man-made impoundments) have dams at one end and shallow tributary inflows at the ends of sometimes numerous 'arms' of the lakes. Figure 1.2.1 illustrates the cross-sectional differences including detail of one hydroelectric dam. This is one common characteristic of reservoirs.

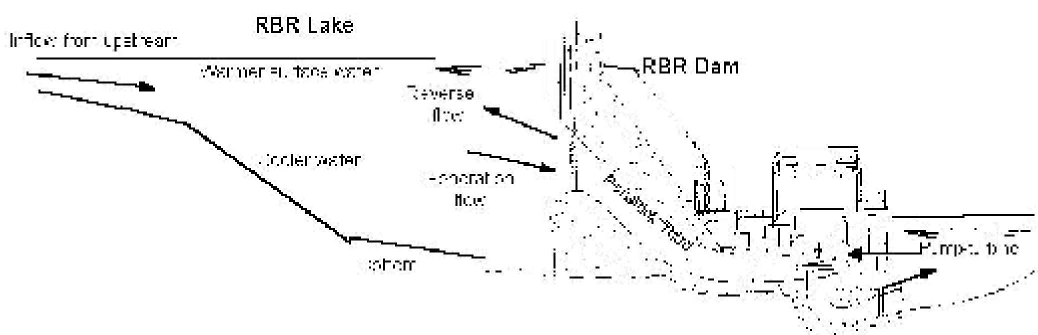


Figure 1.2.1 Cross-sectional diagram of Richard B. Russell Lake showing the dam detail and the placement of intakes and outlets

In the United States, reservoirs are distributed differently from natural lakes. They are often located in areas of water demand or frequent flooding and large reservoirs are rarely constructed in areas where natural lakes are plentiful.

The geographic location and the basin shape of a lake or reservoir are important to the things that happen to and in lakes. The shape of the lake basin is often studied as a topic called **lake morphometry**, the measurement of lake shape. Measurements such as depth and area of a lake are morphometric measurements.

Many reservoirs have shapes that have been termed **dendritic**. Dendritic means that the numerous channels and arms of the lake branch like a tree into smaller channels and arms. This means that for the same surface area, lakes often have much more shoreline than a round lake of similar area. Lake Keowee, a Duke Power Co. reservoir in South Carolina, has an area of approximately 74.4 sq. kilometers. Lake Keowee has approximately 480 kilometers of shoreline although a round lake of similar area would have only a little more than 30.5 kilometers. As you can see, Lake Keowee is not only more than 10 times as complicated as a round lake, ten times as much shoreline is available for housing, docks, beaches, septic tanks, and boats. The impact that this development has on ordinary lakes is increased where more people share the same quantity of water. Table 1.2.1. summarizes the differences between natural lakes and reservoirs.

1.2.5.3 Physical Processes - Light and Thermal Energy

Source and Fate of Light Energy. All life, all ecosystems are dependent on solar energy. The amount and quality of this energy is dependent on the latitude on earth, local climate, altitude, and the season. The fate of this energy in the aquatic ecosystem is dependent on the optical characteristics of the lake or reservoir.

Almost all of the energy from the sun arrives on the earth as electromagnetic radiation. The complete spectrum of this ranges from wavelenghts associated with X-rays (10 nm or less) to wavelengths extending through microwaves and into the radio wavelengths (10's of meters in length). According to Herman and Goldberg (1978), more than 99% of the energy exists in a range of approximately 275 to 5000 nm. For the entire solar spectrum, the integrated average rate of this energy input is termed the "solar constant" or approximately 1.353 (10^3) Watts per sq. meter. The atmosphere, the earth-sun distance and other factors result in significant variations of this 'constant' that can and do affect climate (Herman and Goldberg 1978).

Solar inputs of energy are considered as factors independent of lake processes, reservoir operations, or most other terrestrial interventions on the ecosystem. As shown in Figure 1.2.2, solar energy has its peak intensities in the visible wavelengths. We see these as colors ranging from blues in the shorter wavelengths to reds in the longer wavelengths. The actual wavelength of this incoming radiation, moreover, is very important to its fate in aquatic systems.

Chemical constituents of the atmosphere such as water vapor, carbon dioxide, and ozone selectively absorb certain wavelengths of incoming solar radiation. The preferential absorbtion of longer wavelengths by such constituents is the basis for the so-called 'greenhouse effect' and the constituents are often termed 'greenhouse gases'. In the ultraviolet, diminishing ozone is responsible for decreased absorbance of harmful ultraviolet wavelengths. And obviously, all

Table 1.2.1. Some Characteristics of Reservoirs Compared to Natural Lakes

Characteristic	Natural Lakes	Reservoirs
Distribution	Mostly in glaciated regions also near rivers , their floodplains and often associated with karst regions or coastal plains.	Located mostly outside region of glaciation. Mostly in south region of U.S. Often in regions of water resource need.
Shape	Rounder, less shoreline complexity	Dendritic, greater shoreline complexity (shoreline development).
Morphometry	Basin with central deep areas	Basin with deeper areas near one end (the dam)
Drainage area	Smaller ratio of drainage area to to lake surface area.	Larger ratio of drainage area to lake surface area.
Theoretical retention time	Longer, sometimes many years	Shorter, often less than 1 year.
Inflows	Often many smaller order streams	Often dominated by one or a few major inflows (sometimes other lake outflows).
Outflows	More stable, lake surface fluctuations smaller	Releases according to demand schedules, lake surface fluctuations greater
Longevity	Longer	Shorter
Nutrient loading	This depends on the watershed characteristics and the size of the watershed.	
Water clarity	Greater	Lesser, especially near headwaters.

plants in aquatic ecosystems are affected by the distribution and availability of wavelengths in the visible range.

Whereas all wavelengths can act to input energy to an aquatic system, the visible wavelengths are those most important to biological activity. Visible wavelengths represent about one-half of the total energy for the solar spectrum. At the edge of our atmosphere peak energy for the solar spectrum occurs at the blue edge of the visible range at approximately 380 nm. At the surface of the earth, because of the atmospheric intervention, the peak energy occurs between 500 and 600 nm, approximately centered in the visible range (Goldman and Horne 1983).

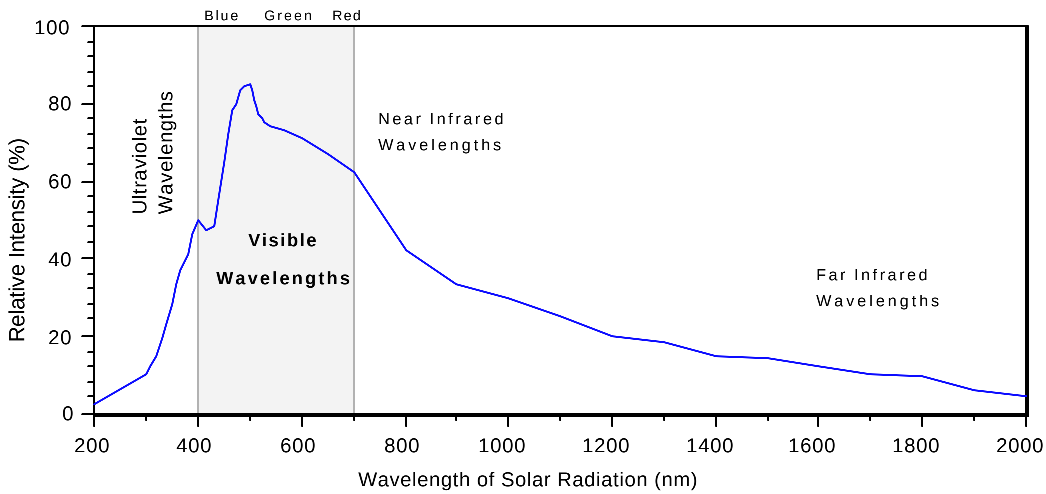


Figure 1.2.2 Wavelengths of Solar Radiation versus Relative Intensity

1.2.5.4 Physical Properties of Water

Water is one of the most important substances for life. Most of the mass of organisms is water and water is necessary for many physiological and biochemical processes. Water is the most abundant liquid on earth and it also simultaneously occurs on earth in solid and gaseous form. It is also almost the only inorganic liquid and the others are not common (elemental mercury, for example).

The physical and chemical properties of water are responsible for the diverse forms and interactions that it is capable of assuming. Water is a simple molecule composed of one oxygen atom combined with two hydrogen atoms. In a sense it is oxidized hydrogen, the product of combustion of organic and other materials. (Water is a product of cellular respiration, for example.) This simple molecule normally would have characteristics similar to those of ammonia or hydrogen sulfide. That is, it would almost exclusively occur in gaseous form at normal temperatures of the earth's surface. However, its remarkable tendency to occur in liquid state is derived from the structure of the water molecule. Oxygen has strong electronegative properties. Its combination with hydrogen results from sharing electrons with the two hydrogen atoms. This is an example of covalent bonding with an inherent assymetry to the bonds. The angle formed between the bonded hydrogen atoms is approximately 105 degrees. This angle is greater than theoretically predicted (90 degrees) because of the repulsive force between the two similarly charged hydrogen atoms. The result is a polar molecule with the negative charge on the oxygen end and the positive charge associated with the hydrogen. Water, then, is a polar solvent. Furthermore, the electronegative pole associated with the oxygen of one molecule may form a weak but significant bond with the electropositive or hydrogen portion of another water molecule, a “hydrogen bond”. This is important for it means that all of the molecules of water in a lake or other body of water are, to an extent, bonded to each other. It prevents the free, independent movement of each molecule that would allow water, for example, to easily evaporate and exist as a gas at normal earth temperatures. Indeed, to form a gas, significant energy must be absorbed to liberate the molecules from the bonding forces.

As water absorbs energy, the motion of the molecules increases and molecular forces bonding the molecules are increasingly strained. The average distance between molecules increases and the density of the water decreases. As energy is lost water cools and the distance decreases while density increases. Maximum density is achieved at approximately 4EC. Figure 1.2.3 illustrates this relationship. As energy is lost and temperature decreases below maximum density, bonding relationships change as water approaches the transition to ice. The structure of ice is rigid. As is easily demonstrated, an aqueous ice bath at standard pressure will theoretically remain at 0EC even with inputs of energy - until the ice has melted. This property is often utilized during instrument calibrations for temperature. The energy exchanged during the conversion of water to ice (or the reverse) without a change in temperature is the energy associated with the phase change and is termed the “latent heat of fusion”. For pure water at standard pressure, the latent heat of fusion is approximately 80 calories per gram. (From the table of conversions, 1 cal. equals the amount of energy necessary to increase the temperature of 1 gram of water by 1^{o} C.) Upon freezing, the rigid crystalline structure of water increases the distance between molecules and density is significantly decreased.

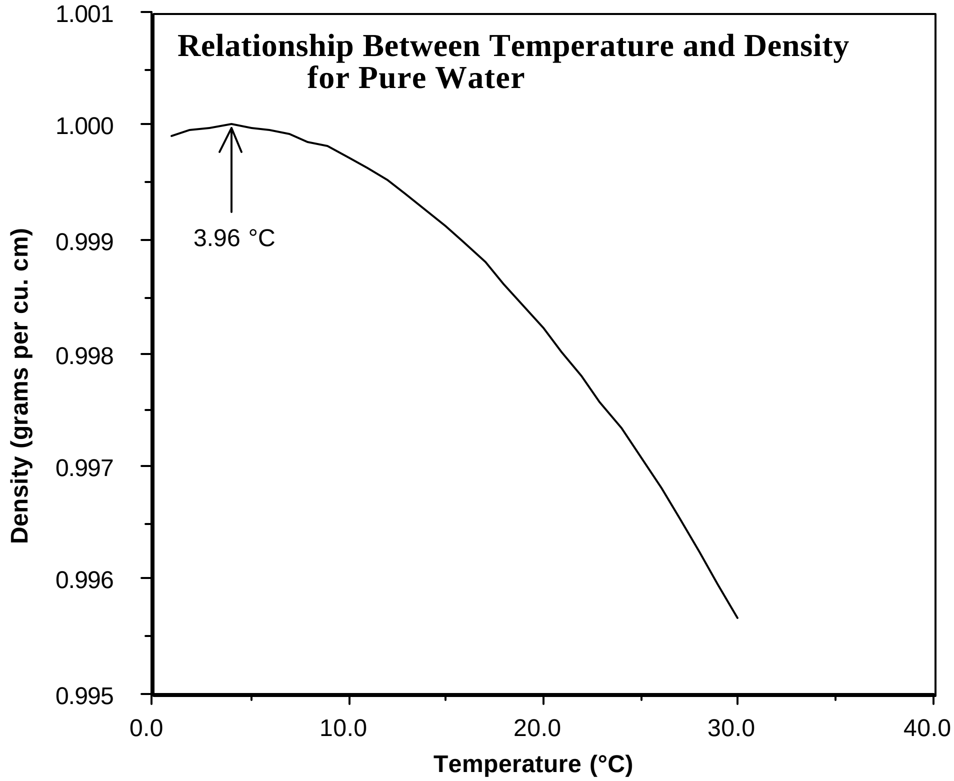


Figure 1.2.3 The relationship between temperature and density for pure water. This is true for most lakes. Exceptions exist where density is influenced by salinity or other chemical concentrations.

At the other phase change, liquid into gas, energy is again absorbed at relatively constant temperature until all liquid has been vaporized. As with the heat of fusion, the temperature at which this process takes place is dependent on the pressure expressed by the environment. At standard pressure this temperature is the boiling point, 100EC, and the energy associated with the conversion of water to vapor (or the reverse) is the "latent heat of vaporization". In this case the amount of energy required for the conversion of water to vapor is quite large, approximately 540 calories per gram of water. This is important for the energy or thermal characteristics of lakes as will be explained later. In the atmosphere this is also important to lakes because the conversion of atmospheric vapor to rain releases this energy. The energy is then available, for example, for expression as large pressure differentials resulting in winds, storms, or even hurricanes.

Between these temperature bounds, water has several interesting behaviors and characteristics. The "specific heat" of water is 1 or unity, defined as 1 calorie of energy required to increase 1 gram of water by 1EC. Density, as shown earlier, is dependent on temperature in a nonlinear manner. Density is also greatly affected by dissolved substances and most persons are aware that salinity increases density as well. If the maximum density of pure water is, by definition, 1 gram per cu.cm then any condition (salinity for example) that increases the density beyond this value will cause the water to 'sink' or displace more buoyant water. Temperature alone, then, would be insufficient to cause further displacement or mixing of that water mass by an overlying mass of different chemical content (less salinity).

The bonding characteristics of water molecules and temperature also affect the property called "viscosity". Viscosity decreases in a slightly nonlinear manner with increasing temperature. This is important for any process involving kinetic motion because viscosity is a measure of the internal friction, the resistance to free, independent motion of water molecules.

1.2.5.4 Thermal Energy and the Ecosystem

Thermal energy is very important to ecosystems. We sense thermal energy by comparison (as in hot water versus cold) or by direct measurement with a thermometer. By knowing the effect energy has on water temperature, we can calculate the amount of energy in a mass of water by measuring its temperature. Often the resultant amount of energy is erroneously referred to heat. All water contains thermal energy, even water that we think is quite cold.

The intensity of thermal energy is sensed by us as **temperature**, but it also changes the water. Colder water usually has greater **density** than warmer water. This means that a volume of cold water usually weighs more than the same volume of warmer water, although very cold water can weigh less than slightly warmer water. Figure 1.2.3 illustrates this relationship graphically.

Because warmer waters are less dense they are more buoyant and tend to remain at or near the surface of a lake. In early March 1997, many southeastern lakes were already beginning to warm in response to increased inputs of light energy. In Lake Hartwell, for example, surface waters had already warmed mensurably more than waters deeper in the lake. This was true for all of the other large reservoirs in the region. Soon thereafter, these lakes were warm enough to remain stratified. From the end of March to the end of April, the temperature of the lake surface increased and the surface temperatures were warmer than the deeper waters.

In summer, surface temperatures are very warm, perhaps 27-30EC. At that time, a swimmer will still be able to easily reach colder water by submerging a few feet below the surface. Most of us are already aware of this. However, the way that the temperatures are distributed are important to all ecological processes in the lake.

During the summer, temperatures in large monomictic lakes decrease from surface to bottom. In this condition the lake is said to be **stratified**. That is, the lake is divided into a series of temperature layers or strata from surface to bottom. Each temperature layer or strata can be identified by locating the depth for each part of the lake at which a certain temperature occurs. Some of these thermal layers or strata are similar to each other in temperature and quite different from other layers or strata.

Limnologists often divide these strata into three general thermal regions of the lake. Figure 1.2.4 illustrates a theoretical stratification pattern for such a lake.

Near the surface of the lake in the summer, temperatures decrease slowly at first with depth. This part of the lake usually is related to the depth to which most light penetrates. This top region of a stratified lake is called the **epilimnion**. Although water temperatures in this layer decrease slowly with depth and sometimes are similar to each other (after a storm, for example) the temperatures of strata at deeper depths decrease more rapidly. The region of rapid temperature change is called the **thermocline**. The depth region of the lake encompassed by the thermocline is often called the **metalimnion**. The thermocline or metalimnion separates the epilimnion from the deepest region in which temperatures again change slowly with depth. The region of the lake deeper than the metalimnion in which there is slow change of temperature is called the **hypolimnion**. Regardless of what else might happen in these regions, they are classically defined by temperature.

The distribution of thermal energy is the consequence of the absorbance of solar energy and any water movements responsible for its further distribution. Figure 1.2.5 shows the theoretical extinction of light in pure water. This pattern is not dependent on wavelength although different wavelengths are absorbed more rapidly than others. This fact is illustrated in Figure 1.2.6.

Actually, we use temperature to define these regions of the lake because temperature is easy to measure. The real reason that these layers are stratified is because of the density differences of the water that are imparted from temperature differences. Under these conditions of stratification, the water in the lake tends not to mix between layers and the lake is said to be

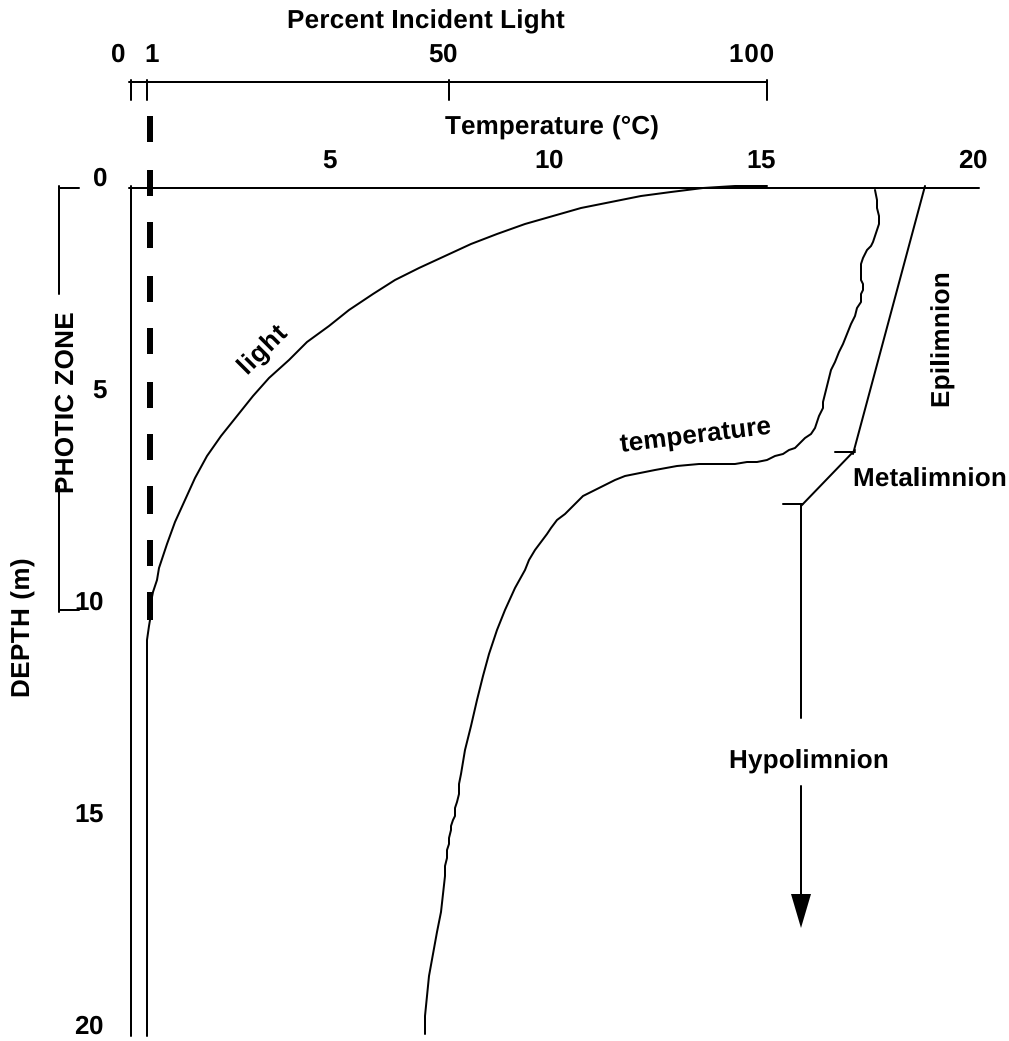


Figure 1.2.4. Diagram illustrating thermal patterns typical of a stratified monomictic lake. The extinction of incident light is also shown illustrating the relationship between absorbance of light energy and the resulting temperatures in a lake.

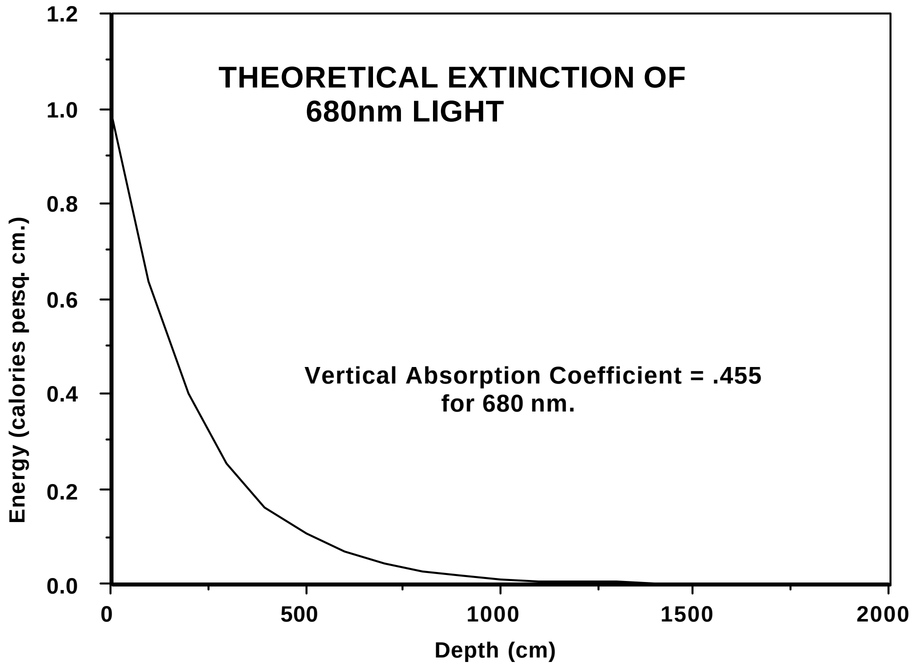


Figure 1.2.5. The extinction of light in pure water. This extinction pattern may be altered where other light absorbing substances are dissolved in the water.

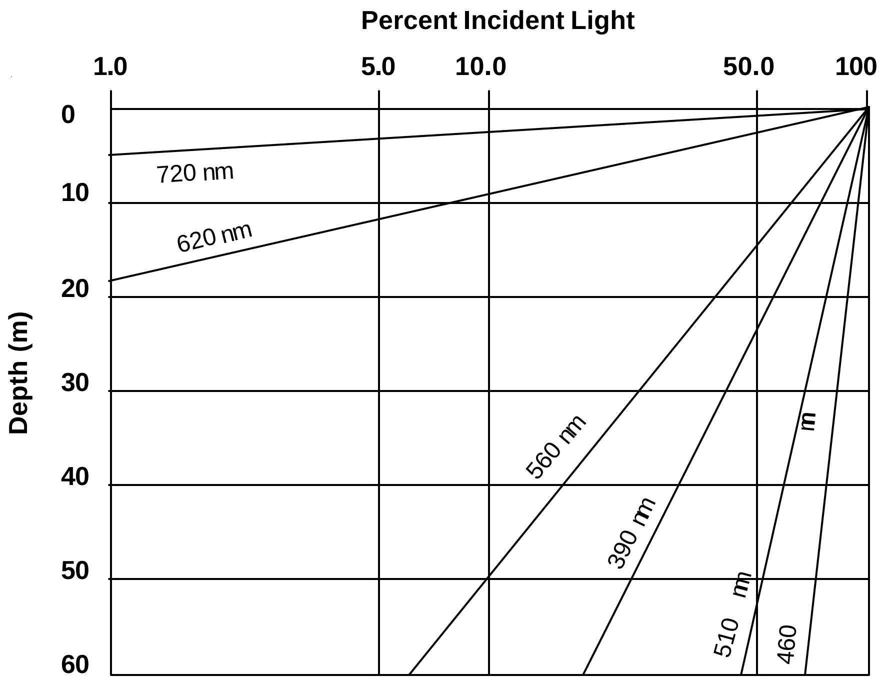


Figure 1.2.6. Wavelength dependence of extinction coefficient and the effect of penetration of light in water

stable. The degree of **stability**, in turn, depends upon the strength of the density gradient or the size of the density difference throughout the water column. Stronger stratification will accompany greater stability. If weather conditions cause surface cooling, however, then denser waters at the surface will tend to be displaced by more buoyant waters from greater depths. This condition of thermal or density **instability** often accompanies **convective mixing** of the waters if winds are not present. Note here that water movement is also affected by its viscosity and viscosity, in turn, is influenced by temperature as well (Figure 1.2.7).

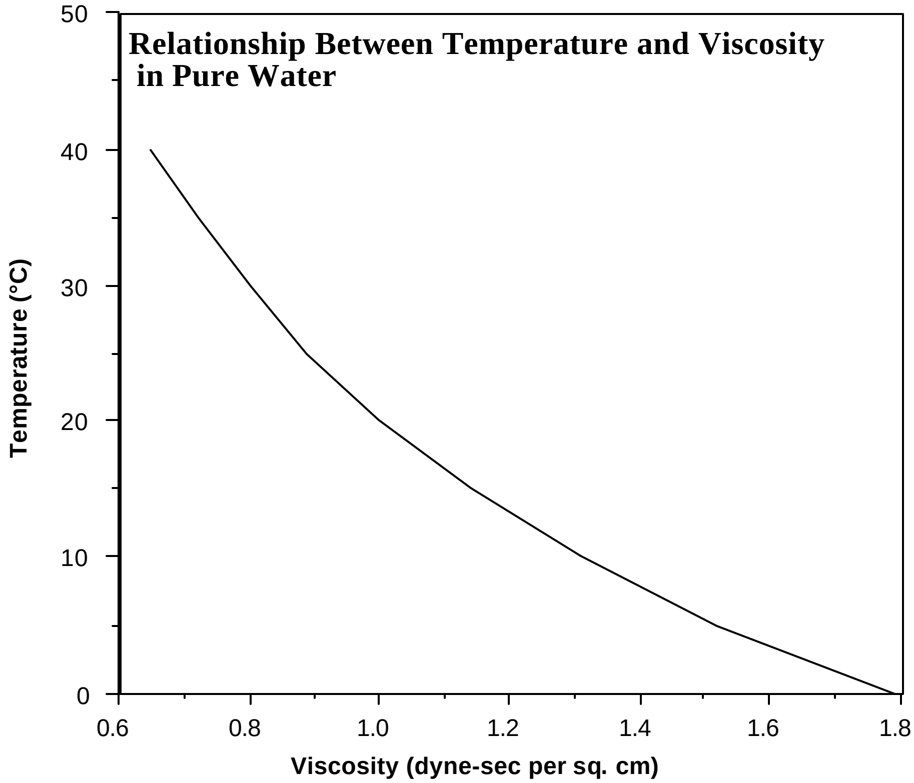


Figure 1.2.7. The relationship between temperature and viscosity for pure water

1.2.6 MIXES

The physical properties of water contribute to stratification, stability and energy budgets in lakes. Because water warmer than 4EC is less dense, warmer waters usually exist nearer the surface. But because water colder than 4EC is also less dense, some temperate lakes have colder waters at the surface (mostly during the winter months). Under both these conditions, the lakes are thermally stratified and considered stable.

In the Southeast, thermal stability usually begins in the spring and continues into the fall. During the fall Lake Hartwell, for example, cools at the surface and experiences cooler inflows. The total energy content of the lake decreases. At the lake surface, the cooling produces shallow instability and wind or convective mixing can occur at progressively greater depths. In Lake Hartwell, this process begins around September and continues through December. When the lake finally achieves a uniform temperature, surface to bottom, through cooling and mixing, the lake is said to be **isothermal**.

In Southeastern lakes, this condition occurs in the winter and continues through additional cooling and mixing until spring. These lakes experience **one season of mixing each year** and are termed **monomictic lakes**.

Water density is not only determined by temperature but also by dissolved substances in the water. The best example of this is salinity which increases water density. In lakes, salinity usually results from activities in the watershed such as winter salt treatment of roadways. But other dissolved substances can contribute to density. When the density of deeper waters of a lake is great enough that complete mixing cannot occur, this condition is classified as **meromixis**. Carters Lake in Georgia and Laurel River Lake in Kentucky are examples of meromictic reservoirs. Neither of them ever experience complete mixing. And the deepest waters of these lakes remain isolated from the atmosphere. This condition can result from chemical and biological processes or from external addition of materials. For many of these lakes, however, their morphometry is such that their depth (especially their mean depth) is large compared to their surface area. There is a morphometric measurement that quantifies this relationship and it is called **relative depth**. Lakes with greater relative depth can achieve meromixis with greater ease.

Many lakes farther north have thermal stratification during the winter. Their surface temperatures will be at or near freezing and often the lakes will be covered with ice. This is because, as mentioned earlier, at very cold temperatures the water becomes a little less dense. In those lakes at those times, the bottom waters will be slightly warmer (and more dense) than the surface.

Because these lakes mix in the fall until they experience winter stratification and then mix again during the spring thaw until they restratify, they have **two seasons of mixing each year** and are termed **dimictic lakes**.

Although this condition may occur infrequently during the winter in small ponds or in shallow isolated areas of coves of larger lakes in our area, it occurs only temporarily and is not a dominant characteristic of Southeastern lakes.

1.2.6.1 Stability and Heat Budgets

Exchanges of energy are represented by such processes as light absorbance, backradiation, as well as the energy exchanges due to inflows and outflows. These exchanges are greatly influenced by lake morphometry and hydrology. Limnologists have incorporated the major features of both of these influences into two measures derived from these incorporated factors. These are stability, a measure related to the effort needed to mix the lake as discussed previously, and the heat budget, a conceptual approach to accounting for all inputs and losses of thermal energy.

Stability is derived from knowledge of the morphometry of a lake and the thermal (actually density) distribution. Simply, positive stability indicates that external force must be applied in order to mix the lake to an isothermal condition (actually uniform density). Zero stability is usually encountered during isothermal conditions. Negative stability indicates that the rate of energy loss from near the surface is sufficiently great that mixing may proceed spontaneously due to the buoyancy of underlying waters. Negative stability therefore is seldom encountered on a large scale and it is usually a short-term phenomenon.

1.2.6.2 Stability Equation

$$S = \frac{1}{A_0} \int_{z_0}^{z_m} \left[\rho_z - \bar{\rho}\right] A_z \left[z - z_{\bar{\rho}}\right] dz$$

WHERE:

S = stability,
A_0 = Area of the lake surface (depth = 0),
A_z = area of lake at depth
z_m = maximum depth
z_0 = lake surface
z- = depth at which water temperature corresponds to average density
ρ_z = density at depth
$\bar{\rho}$ = volume-weighted average density

Heat budgets may be derived using two different approaches. The simplest approach is to employ morphometry with known temperature distributions (which are used to calculate densities in the stability calculation, see the relationship?) This is an empirical approach because it uses knowledge in hand to further describe characteristics which already exist. Using known specific heats for water and the temperature and the depth-volume relationships, it is simple to calculate the amount of thermal energy in calories or other units contained in the various depths of the lake. These are then summed and the total energy (caloric) content can be calculated relative to some reference value. This reference value is usually chosen as the minimum temperature experienced by the lake during the year. Calculating the changes in heat content, it is easy to see the effect of various processes on the entire lake as the seasons progress.

Another approach to the heat budget is more complex but this is the approach used in many sophisticated models. This analytical approach uses the rates of energy exchange combined with lake morphometry to predict the outcome of any modification to energy exchange processes. It can also predict the same characteristics that can be calculated empirically using the method previously described. Obviously, this is the more powerful approach and these models are very successful in making useful, accurate predictions about thermal characteristics.

1.2.7 RESERVOIR ZONATION

Water moves laterally in lakes as a result of inflows and outflows. Reservoirs may be functionally divided into three regions in which these flows have different characteristics. Headwaters are often dominated by processes similar to and as a result of the riverine inputs to the region. If inflows have a density greater than lake surface waters, the inflows will tend to **plunge** beneath the lake surface. Often a trash line of floating debris will indicate such a plunge point. If the inflow water is less dense, it will flow over the lake surface. If inflows are greater density than the lake surface but less dense than lake bottom waters, they may form an **interflow** which extends some distance into the lake or perhaps throughout the lake. Such interflows are common where plunging inflows attain depths similar to the penstock opening depth on the dam impounding the lake. In all instances, substantial inflows can greatly influence thermal structure in the lake.

The region of a reservoir where the inflows dominate the lake characteristics is considered the **riverine zone** of a reservoir. In this region, the types of processes occurring are more like a river than a lake.

In the deeper regions reservoirs are often dominated by processes typical of open-water (limnetic) environments. The region in which the lake gradually changes from riverine to limnetic dominance is aptly termed the **transition zone**. Care must be taken when identifying these zones in lakes because lake levels, flows, seasonal thermal and chemical processes can vary in ways to cause the positions of these zones to change. This is especially true of the transition zone.

In the deepest region downstream from the transition zone and where strictly limnetic processes dominate is the **lacustrine zone**. This zone extends to the dam in reservoirs. The three zones often change size or position depending on the trends of rainfall and resulting flows. For example, high flows from occasional storms may extend the transition zone additional kilometers downstream in West Point Lake. And in Richard B. Russell Lake, reversed flow with pumped storage gives the forebay lacustrine zone properties similar to the transition zone. Increased flows during the winter can greatly diminish the lacustrine zone and greatly enlarge the riverine and transition zones.

Inflows are important sources of materials to lakes. Materials input from external sources are termed, **allochthonous** materials. Some materials can have their source internally (oxygen and carbon, for example) and when they have an internal source they are termed, **autochthonous** materials. These terms apply to all materials whether they function as sources of turbidity, nutrients for plants, or food for fish and other animals. You will be tested on your ability to spell these two terms.

1.2.7.1 Chemical Processes

A budget accounting for the inputs and outputs of materials can be constructed in a manner similar to the energy budget or the household budget. In some ways chemical budgets are actually simpler (depending on the material) and questions related to material budgets are often related in terms of material **loading** to the lake. 'The term 'loading' correctly implies that there is an accumulation of some material of interest that is being transported into the lake from some external source, most often streams or seepages. In most reservoirs, loading is a concern because reservoirs are often located in watersheds having activity which contributes allochthonous materials of interest to the inflowing streams. Among these materials are sediments, organic materials, toxic materials, and the nutrients. Only the nutrients will be discussed in this workshop.

Reservoirs are very sensitive to loading because of the proportionally larger watersheds compared to natural lakes. The greater watershed area and resulting inflows of materials are often trapped by reservoirs. Figure 1.2.8 illustrates the effect of construction of Richard B. Russell Lake on phosphorus loading to J. Strom Thurmond Lake, another reservoir just downstream. The phosphorus loading by the Savannah River prior to Richard B. Russell Lake was greatly decreased as a result of the impoundment of Richard B. Russell Lake. Another major tributary to J. Strom Thurmond Lake is now the major source of phosphorus to that lake although the amount of water entering from the Savannah River has not decreased. Furthermore, because much of the phosphorus leaving Richard B. Russell Lake is dissolved phosphorus, the form of this nutrient loading has been changed as well.

The two variables that must be measured to assess inputs of materials are streamflow and concentration. These may then be used to compute actual quantities of materials entering an aquatic system. Concentrations may be measured chemically, flows are measured using some type of gage. The time-dependent record of flow for a stream is termed a **hydrograph**. Figure 1.2.9 depicts a theoretical hydrograph showing base flow and storm flow for a hypothetical

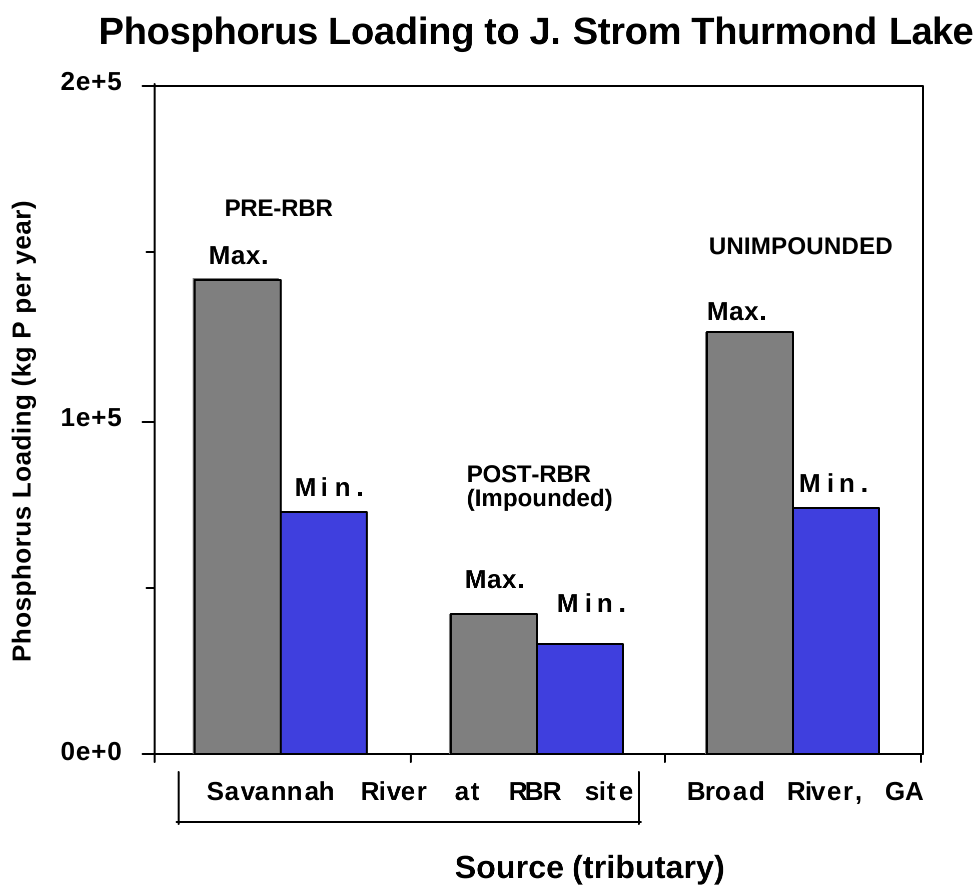


Figure 1.2.8 Consequence of construction of Richard B. Russell Lake on phosphorus loading to J. Strom Thurmond Lake, just downstream

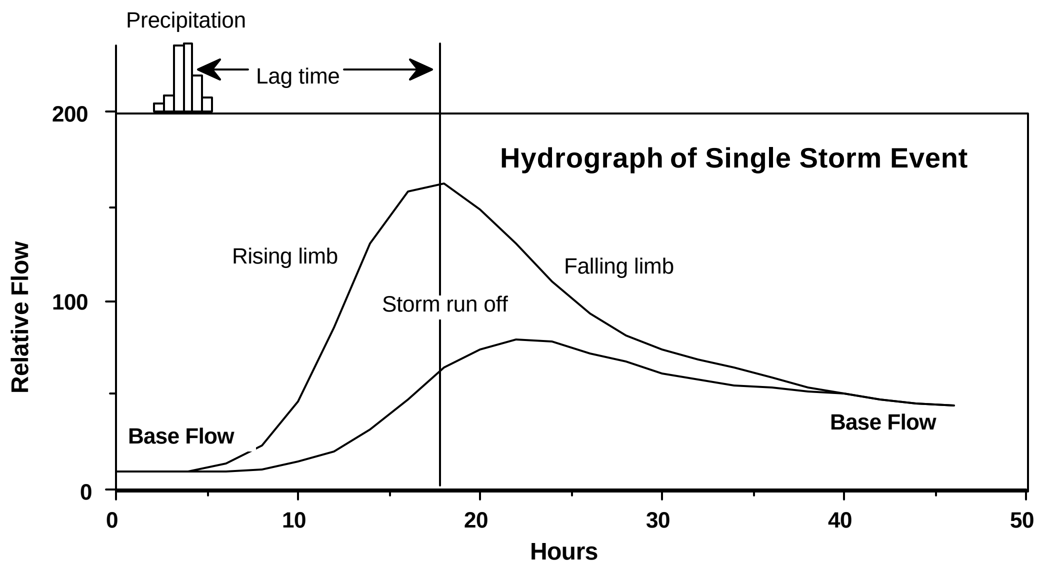


Figure 1.2.9 Theoretical hydrograph showing the major features of stream flow through a storm event

stream. A large quantity of water (and water-born materials) may enter a lake via streamflow without detection if the brief period of the storm hydrograph passes unobserved. Moreover, the concentration of materials is rarely uniform throughout these flow changes. Often the maximum concentration will occur early in the hydrograph, perhaps before maximum flow is attained. For this reason, measuring loading to lakes can be very intensive work - especially if there are numerous significant inflows.

1.2.7.2 Important Chemical Elements

Assuming that a lake already contains materials in the water, there are important processes which influence their distribution and form. The materials of greatest interest in reservoirs are Oxygen (**O**), Carbon (**C**), Nitrogen (**N**), Phosphorus (**P**) (these are four macronutrients) and the metals, Iron (**Fe**) and Manganese (**Mn**) (sometimes Mg, Na, K, Ca and others especially if toxic materials are suspected). Of all of these, three (Oxygen, Carbon and Nitrogen) exist in both gaseous and chemically-combined solid forms. And to better understand these three elements, laws governing gases must be understood. Technically, hydrogen (**H**) is also a major element because it is one of the two elements in water and also exists in biological tissues along with carbon and oxygen in the form of biochemicals such as **carbohydrates**. However, hydrogen does not normally exist as a free gas and is almost always combined with other elements.

The ideal gas laws apply approximately to the major components of air, Nitrogen, Oxygen and Argon. Under most circumstances, argon is ignored because it composes less than 1% of the gaseous content and because it is an inert gas. Because it does not enter into complex reactions, it is a **conservative element**.

Nitrogen gas also enters few reactions (nitrogen fixation is an exception) and under some conditions may be treated as an inert gas. However, nitrogen combined in other compounds is not conservative and enters a large number of biological and chemical interactions. Nitrogen is an important element to biota. But as a gas it is available to biota only under the extraordinary circumstance of nitrogen fixation by procaryotes. (Don't worry, that topic will come later). Otherwise Nitrogen gas is approximately conservative and does not enter biological processes. As a gas nitrogen is important mostly when there is too much of it (when it is **supersaturated**). Supersaturation can cause injury or death to fish and other biota and this is a topic of tremendous interest in the Pacific Northwest or other regions where the phenomenon impacts wildlife resources. This is not a common problem in the Southeast but supersaturation has harmed fish at some hydroelectric facilities.

Oxygen is obviously non-conservative. However, under very small time scales oxygen can occasionally be viewed as approximately conservative. At such short time scales, the reactions affecting oxygen concentrations do not proceed quickly enough to significantly affect the process. Carbon dioxide forms only approximately 0.033% (and increasing) of the atmosphere and is not a conservative material in water. Instead it enters chemical reactions that allow it to take forms not typical of gases.

Oxygen is one of the most important gases in a lake. Its presence or absence determines not only the type or abundance of fisheries and other biota, but it also determines indirectly the type of chemical processes that may take place in the lake. Oxygen is extremely chemically and biologically reactive and is not conservative. Its distribution in a lake is determined by its solubility and lake conditions.

1.2.7.3 Henry's Law of Solubility

$$c = K\,p$$

Where,

c = the concentration of gas absorbed at equilibrium
K = the solubility factor
p = the partial pressure of the gas.

This simple relationship controls the solubility of gases in water. It is an equilibrium expression and as such makes an assumption that equilibrium has been or will be attained. The **solubility factor** is difficult because not only is it unique for each gas, but it varies with temperature and the presence of other dissolved substances in the water. The **partial pressure** also varies with the environment. Water at equilibrium with one atmosphere of air pressure (the major source for dissolved gases in lakes) will have dissolved oxygen with a partial pressure of approximately 159 mm Hg (20.9% of atmosphere times 760 mm Hg atmospheric pressure). Pure oxygen delivered to water from a diffuser head submerged to a depth equivalent to 4 atmospheres of pressure (approximately 40 meters) will have a partial pressure of 3040 mm Hg at equilibrium (*don’t be frightened by this stuff*).

Gas dissolved in water still exerts a pressure **and at equilibrium will exert the same pressure as the fraction in the gaseous phase**. This means that if the water in a lake completely absorbs atmospheric gases and attains equilibrium with the atmosphere, then the pressure of oxygen in the air will be equal to the pressure of oxygen dissolved in the water. Even though the concentrations are quite different. It's that simple.

In the presence of thermal gradients and biological processes which tend to remove oxygen through respiration, oxygen may be distributed in a variety of ways. A **clinograde** distribution occurs when oxygen concentration continuously declines with greater depth. With sufficient mixing, the distribution of oxygen may be uniform, surface to bottom. **Positive** and **negative heterograde** distributions occur when oxygen departs from the clinograde trend. These terms have less meaning today than they did when lakes were classified according to the types of material and thermal distributions but these terms are still occasionally used by limnologists and aquatic ecologists.

Carbon dioxide is an important product of respiration and enters a chemical reaction system called the **carbonate equilibrium system**. At equilibrium for this system, inorganic carbon will exist in several forms dependent on pH.

$$pH = -\log [H^+]$$

pH is related to how acid or alkaline water is and it is one of two master variables for determining chemical equilibria. (*Don't let this scare you either*) At equilibrium, carbon dioxide will exist in water at pH less than approximately 4.3 as the dissolved gaseous form and the hydrated form, carbonic acid (H_2CO_3). At pH greater than 4.3 carbon dioxide may take ionic forms, bicarbonate (HCO_3^-)and at even greater pH values, carbonate ($CO_3^=$). The carbonate ion is not a major form of inorganic carbon for most lakes and it is unavailable to plants as a source of carbon. Please refer to the accompanying illustration for the relationships of pH to the forms that carbon dioxide takes in water.

Metals may be dissolved in water in varying concentrations depending on the source, the pH, and the presence or absence of dissolved oxygen. If oxygen is absent, then a chemical condition called a **reducing** environment might form. This is in opposition to an **oxidizing** environment (see the connection to oxygen?). Anaerobic environments may or may not be reducing ones but aerobic environments are nearly always oxidizing ones. Chemically reduced conditions allow the dissolution of certain metals, especially iron and manganese. These elements are important micronutrients for biota but can be toxic in high concentrations.

Other important metals include the alkaline earth metals such as calcium, magnesium, sodium, and potassium. Calcium is an important nutrient for plants and animals, especially those forming calcareous shells. Insufficient calcium may greatly affect distributions of clams and mussels, for example. Calcium also affects alkalinity and hardness of water.

Magnesium is important for chlorophyll synthesis in photosynthesizing plants. Each chlorophyll molecule contains one atom of magnesium. Magnesium also affects alkalinity and hardness and is (with calcium) one of the most important divalent cations (positively charged ions with a charge of 2+, Ca^{2+} or Mg^{2+}, for example).

Sodium (Na^{+}) and potassium (K^{+}) form monovalent cations and are also common chemicals in water. Potassium is an important nutrient element for plants and animals. Sodium, however, does not enter many reactions and can often be used as a conservative tracer for describing water movements.

During stratification, oxygen may be depleted in the hypolimnion forming an anaerobic zone. In that zone, chemical reduction may allow iron or manganese (or methane and sulfides) to be produced in or from the sediments. Such processes are common in reservoirs. Reservoir releases from penstocks (openings in the dams that lead to the turbines) that withdraw from those depths will contain reduced iron and manganese. If placed in an aerobic environment, the subsequent oxidation of iron is rapid. But manganese oxidizes slowly and can remain in the reduced state long distances downstream even in aerobic environments.

It is important that even if we can predict an outcome at equilibrium, the system rarely ever actually attains such equilibrium and predictions must take kinetics into account. This is one reason that chemical environmental models do not enjoy the degree of prediction accuracy that physical models enjoy.

Oxygen depletion rates can be calculated using monitoring data collected over time. Such calculations are useful as tools for predicting anoxia or decreased concentrations of oxygen in outflows. Another way to measure an oxygen depletion rate is to use bottles or other enclosures to measure oxygen loss directly.

If one reservoir releases to another reservoir, processes in one lake can affect processes or distributions in the other lake. This is related to loading and is not confined to metals or nutrients. Such interactions are illustrated by the historical distributions of September dissolved oxygen concentrations in J. Strom Thurmond Lake. After the impoundment of Richard B. Russell Lake (RBR Lake) in 1984, there was a trend of greater depletion of dissolved oxygen in J. Strom Thurmond Lake.

Variation over long time scales is a problem that only long-term monitoring programs can assess. The long term effect of impoundments can sometimes be identified or predicted soon after impoundment. For example, Richard B. Russell Dam altered the flow of the Savannah River and decreased the nutrient loading to Clarks Hill Lake. The amount of water was undiminished but settling of particles in inflows to RBR Lake decreased the amount of total phosphorus entering Clarks Hill

Lake. The long term effect of this decrease in available nutrients would require long-term study and careful consideration of changes that may take place elsewhere in the watershed for Clarks Hill Lake.

1.2.8 HABITAT AND LIFE ZONES

Habitat, as most people already understand, is often viewed as the place, space or environment in which an organism lives. Many of our management concerns are for habitat. Habitat is often confused with **niche**, a concept that is much more abstract and is defined as characteristic of the organism or population. Eugene Odum compared the niche to an organism's profession and the habitat to its work environment. In aquatic systems, habitat is often defined or bounded by physical and chemical properties of the lake or stream. In lakes these properties include depth, temperature, light intensity, light quality, dissolved oxygen and nutrient concentrations, and sometimes other chemical concentrations or factors such as oxidation-reduction potential or pH. Because some of these habitat boundaries coincide closely with the physical and chemical limnological zones discussed earlier in the course, terms that can be applied to most lakes exist for these habitats.

The epilimnion of a lake is often similar in size and limits to the **photic** or **euphotic zone**, a zone which exists from the surface to the depth of approximately 1% of surface light intensity. This correspondence should not be surprising if one considers the effect of light absorbance on temperature and thermal energy distribution in a lake. The euphotic zone, however, can be affected by many factors which alter the absorbance of light, organisms included. And while the euphotic zone is primarily important for plant life in lakes, it is also important to animals whose feeding and reproductive behaviors depend on specific light environments. The euphotic zone is highly variable spatially and temporally, light being quicker to respond to external influences than temperature.

The euphotic zone has been defined by the depth of 1% of incident light and this rather arbitrary limit betrays an early belief by limnologists that plants could not effectively utilize light intensities less than 1% of surface illumination. At lesser depths photosynthesis could actually add dissolved oxygen to the lake. Numerous exceptions to this generality are know today.

Surface illumination is dependent upon many factors, latitude, date, time-of-day, climatic conditions. The **solar constant**, the constant maximum light intensity arriving to the earth at the outer edge of the atmosphere, is approximately equal to 1.94 cal. cm^{-2} $minute^{-1}$. Some of this light does not enter a lake due to the factors mentioned earlier such as absorbance or reflectance by the atmosphere or water surface. Assuming that only 1% of the incident light (0.02 cal. cm^{-2} $minute^{-1}$) defines the euphotic zone, then **photosynthesis** (the process by which light is used to convert inorganic molecules into organic compounds) at that depth would be assumed not to exceed **respiration** (the process by which organic compounds are decomposed biochemically to yield energy and inorganic molecules). The depth at which photosynthesis was equal to respiration is considered the **compensation depth**, or the depth at which photosynthetically produced food exactly compensates for the loss due to respiration. Today, we know that some organisms are capable of utilizing light intensities less than 1%

of surface intensity. Among these are photosynthetic bacteria, cyanobacteria (blue-green algae) and some green algae.

Where adequate light illuminates the bottom of a lake, rooted vegetation can become established, depending on sediment type and nutrient availability, and the zone where this condition exists along the shore of a lake is called the **littoral zone**. In the open-water, deeper areas of the lake, different organisms often are adapted for this **limnetic zone** or **pelagic zone**. The term ‘pelagic’ often is used for the open ocean areas of the world whereas the term ‘limnetic’ is restricted to use in freshwater environments.

As already discussed, light quality (wavelength) also changes with depth with the general trend of short wavelengths penetrating deeper than longer wavelengths. To compensate for this certain organisms living in deeper waters have developed the ability to preferentially utilize those wavelengths of light that are present. This ability is called **chromatic adaptation**.

The diagram illustrating longitudinal gradients in reservoirs shows the major distinction between impoundments and natural lakes. Most reservoirs have longitudinal gradients in water quality. Natural lakes have similar gradients but those gradients are not always longitudinal nor are they usually as dramatic as in reservoirs. Reservoirs are conceptually divided into three functional zones. The **riverine zone** is located close to major inflows and extends into the **lacustrine** or **limnetic zone** longitudinally until the horizontal (**advective**) influence of the river or other inflow has become less than the influence of lake or limnetic processes which are characteristic of lentic environments. Between the limnetic or lacustrine zone and the riverine zone is a region in which a transition is made between the dominant influences on lake water quality. This is called the **transition zone** and it varies in magnitude and position between lakes as well as temporally within any lake. These temporal variations arise from lake level changes, from variations in river flow rates, from thermal trends (which are often seasonal) and from shifts in sediment load or other loading materials.

Reservoir biota or aquatic organisms often have apparent preferences for certain habitats. This fact is reflected in their distribution among depths or habitat zones in reservoirs. As a group, fish are found in nearly all potential habitats except those such as the sediments or sands. Yet certain types of fish specialize for preferred habitats. As a group the fish belong to the functional classification of **nekton**, organisms which are capable of controlling their own positions and are capable of doing this in spite of water currents. Fish are the major components of the nekton although some insect larvae may be considered as part of this functional group as well. Organisms which are unable to control their position in the presence of water movements are called **plankton**.

The terminology associated with the biota is extensive and diverse. It reflects size, function, habitat and other characteristics. **Benthic** organisms inhabit the bottom of the lake. **Meroplankton** are planktonic for only part of their life cycle and inhabit some other habitat at other times. **Neuston** are organisms, usually microscopic, which associate with the surface film of a lake. They are sometimes divided into **epineuston** and **hyponeuston** (above and below the surface film). **Pleuston** are organisms which live on the surface but penetrate through the surface (duckweed, for example). There are many more such distinctions. Had enough?

These organisms all act as agents of exchange between the physical environment especially with regard to light, and the chemical environment especially with regard to oxygen and nutrients. And the ecosystem perspective is the best way to make sense of the diversity and complexity of these lake systems. In this perspective, materials and energy are viewed as moving, interacting components of a lake, both affected by the organisms and affecting the organisms.

Biota may modify the way materials are distributed in lakes. For example, much of the phosphorus in some lakes is in organismal form. This means that if we removed the organisms (the plankton, the fish, etc.) then we would remove the nutrient as well. It also means that other forms of the material are absent in other parts of the lake (the sediments for example) and that inputs from the watershed or other sources do not significantly exceed the rate of growth of organisms. An example of such a system might be, for example, an oligotrophic lake like Jocassee, a pumped-storage reservoir in the Blue Ridge escarpment of South Carolina. Phosphorus, when it enters such a lake, is rapidly accumulated into biomass. In such situations, these nutrients might be considered the **limiting factor** for biological growth. That is to say, that with all other factors held constant, increasing the supply of the limiting nutrient is all that is required for increasing **production** of biomass.

Functionally, growth of biota is often divided into several categories. **Primary production** is that biomass which is the direct result of photosynthesis. This activity is limited to plants, either phytoplankton or macrophytes, and some photosynthetic bacteria. Organisms that eat primary producers are termed **secondary producers** or **primary consumers**. Many zooplankton and all cows (except, of course those which are fed meat by-products) are primary consumers. So are grass carp, manatees and some insects. Organisms which eat other consumers are termed **predators** or **carnivores**.

This has the semblance of organization and it has been formalized with trophic structure - or who eats whom. And this system is termed **trophic dynamics**. Ecologically, this is important because it organizes how materials and energy are exchanged between functional organismal groups. This system is important to lake management because it allows models of biota to be constructed for prediction of effects of management strategies. Trophic dynamics has no relation, however, to lake trophic status and the distinction is easily shown.

Lake **trophic status** is based on the idea that as lakes age they tend to become shallower in depth, succeeding to less limnetic habitat and more wetland habitat and finally to forest habitat. The idea of lake succession was originally described for northern glacial lakes but applies in some ways to reservoirs as well. A lake may begin as a basin filled with water with low concentrations of nutrients and sparse biota (an **oligotrophic lake**). With time, inflows deposit more nutrients and biota take advantage of this with increasing populations (**mesotrophic**). There is some sediment accumulation. With more time, nutrients may become very concentrated and the lake **eutrophic**. The lake begins to fill in and marshes encroach on the open waters and finally the lake changes to a wetland. There exist reservoirs less than 100 years old that have converted to woodlands. In contrast many glacial lakes have existed for thousands of years without completing this process.

1.2.9 BIOTA: MAJOR COMPONENTS IN LAKES

A rich terminology has been constructed for describing phylogenetic, spatial and functional categories of biota in lakes. Some of the terms are arcane and seldom encountered, others are commonly encountered and usually taken for granted by limnologists.

Lake organisms perform the same functions as organisms in any other ecosystem. Their special adaptations enable them to inhabit the specialized environments found in many lakes. For most purposes, they include plants, animals, and diverse microbes. Spatially, organisms inhabit the surface, the lake bottom, and either float or swim in the lake.

Among all organisms there is one great biological distinction that makes some of the following groups difficult to organize. All organisms are either **eucaryotic** (having complex cell organization) or **procaryotic** (having simple cell organization). All bacteria are procaryotic. All multicellular organisms are eucaryotic. **Cyanobacteria**, otherwise known as blue-green algae are procaryotic. All of the fungi and protozoans are eucaryotic. People are eucaryotic.

Now the terms...

I. Plants

A. **Macrophytes** - large vascular plants, can be observed without microscope.

1. **Endemics** - native to region, *Valisneria, Ceratophyllum*
2. **Exotics** - introduced to area from other places. *Hydrilla,* Eurasian watermilfoil
3. **Pleuston** - plants which float on the surface and penetrate into the water.

B. Microscopic plants - algae, unicellular and eucaryotic

1. **Phytoplankton** - microscopic plants subject to water movements. Reservoirs are dominated by green algae, diatoms, and dinoflagellates (also cyanobacteria)
2. **Neuston**, etc. - organisms associated with the surface film of the water.
3. **Aufwuchs** - attached organisms, often called **periphyton**.

Specialized terms according to type of surface, **epiphytes** on plants, **epilithon** on rocks, **epipsammon** on muds and sediments, **psammon** living interstitially in the sediments. *Aufwuchs also contains animals and microbes.*

4. Filamentous benthic or suspended mats - Filamentous green algae such as *Spirogyra* or cyanobacteria such as *Anabaena* or *Lyngbya*.

The common nuisance algae in lakes were named Annie, Fannie and Mike after *Anabaena*, *Aphanizomenon*, and *Microcystis*, three noxious cyanobacteria noted for forming 'blooms' and surface scums in many lakes considered to be eutrophic. In addition the algae also include numerous groups such as Euglenophyta (*Euglena*), Chlorophyta (*Chlamydomonas, Volvox, Oedogonium, Spyrogira*, *Zygnema*), Chrysophyta (*Diatoms, Mallomonas*), and the Pyrrophyta (dinoflagellates) which inhabit the diverse habitats present in most lakes.

II. Animals - (neglecting waterfowl, mammals, and amphibians)

A. Swimmers - **nekton** - capable of moving against water currents

1. **anadromous** fish - river and stream spawners
2. **catadromous** fish - ocean spawners
3. **pelagic** - inhabiting the open deeper waters
4. **littoral** - inhabiting the shoreline shallows

B. Weak swimmers, **zooplankton** - capable of moving but not against modest currents

1. Copepods - mostly grazers, cyclopoid most common in lakes, example *Cyclops*, *Diaptomus*.
2. Crustacea - grazers and predators, numerous examples including Cladocera (*Daphnia*, *Bosmina*), Ostracoda
3. Rotifera - rotifers such as *Keratella*, often much smaller than other zooplankton, mostly grazers, may be major component of reservoirs
4. **meroplankton** - organisms inhabiting the plankton only during part of life cycle, often as larvae. Clams, insects.
5. **ichthyoplankton** - larval fish

C. Insects - meroplankton

1. Chironomidae - midge larvae often red with haemoglobin, especially in anoxic sediments.
2. Chaoboridae - Chaborus may be most important single insect in some reservoirs. Called phantom midge, has complex migration pattern through the water column on a diel basis.

D. The Benthos

1. insects - includes the above insects as well as Trichopterans (caddisflies), Ephemeroptera (mayflies), and others.
2. worms - Oligochaeta, true worms. Often indicators of undesirable sediment qualities when found as dominant form.

3. Molluscs - filter feeders, clams and mussels, *Corbicula*, zebra mussels, and deposit feeders, snails.
4. benthic crustacea - Ostracoda, Cladocera (*Sida*), crayfish, etc.
5. Other groups - mostly filter feeders, bryozoans (*Pectinatella*), sponges (*Spongilla spongillis*), Hydrozoans (*Craspidecuspa*).

III. Microbial assemblages - (except algae)

A. Bacteria - functional classes
1. Photosynthetic bacteria (not cyanobacteria)
2. sulfur or metal reactive bacteria, *Metallogenium, Chlorobium*

B. Fungi - the Phycomycetes, *Saprolegnia, Lagenidium*

C. Protozoa, Vorticella, Stentor, Paramecium, Amoeba - sometimes deadly parasites (*Acanthamoeba, Naegleria*), or merely parasitic (*Epistylis* on fish).

Such classifications are promoted as simplifying and organizing the great diversity of aquatic life. These designations are never perfect, however. The plankton, for example, contains microbes, phytoplankton and zooplankton, and a large variety of meroplankton, larvae that temporarily inhabit the plankton. The aufwuchs (periphyton) also contains microscopic plants, animals, bacteria, as well as worms, and benthic insect larvae. At some very fine scale, the organizational relationships become unmanageable except within the context of individual life histories.

1.2.10 BIOTA 2: FUNCTIONAL RELATIONSHIPS

Here we concern ourselves with what the organisms do. And it is here that the ecologists have constructed elaborate predictive systems to organize the apparent chaos that we discovered in the previous section.

1.2.10.1 Distributions

Vertical distributions and longitudinal distributions through reservoirs are greatly affected by stratification and flow patterns. During mixed conditions plankton are often well-distributed throughout the depths of a reservoir. While the lake is mixing, cells are periodically brought into contact with the lake surface where light is available and the motion of the water causes greater contact with nutrients. Such conditions are favorable for productivity. For this reason, the early spring and the fall are often times of increased planktonic biomass.

As stratification begins, heavier particles tend to sink and settle out of the water column. Diatoms and larger planktonic cells commonly do this. In the northern natural lakes there was a trend for the spring diatoms to be replaced by greens and cyanobacteria. This was presumably due to competitive interactions for phosphorus and silica (which is needed by diatoms). The idea was formalized by Claire Schelske and Gene Stoermer as a hypothesis to explain the succession of plankton in the Great Lakes and a response to eutrophication in Lake Michigan.

We now know that under stratified conditions, vertical zonation of processes and biota occurs in reservoirs. The photic zone is inhabited by planktonic organisms and fish. Sometimes a layer of organisms accumulates near the thermocline and this is called a **metalimnetic layer**. Water density and viscosity changes interact with the organism densities and shapes to help form these layers. The layers can be sites of production as well as mineralization of organics.

Longitudinally, production in a large reservoir tends to peak around the region of the transition zone, diminishing toward the headwaters as they become more riverine and toward the lacustrine waters of the forebay as they become more depleted of nutrients as well as suspended materials. However additional water movements such as **upwellings** in the lacustrine zone can reverse this trend and cause increased production.

1.2.11 RESERVOIRS

Reservoirs are the product of a manmade structure which was constructed for specific purposes resulting in the control of the hydrology of the river. These purposes include flood control, hydropower generation, water supply, navigation, low-flow augmentation, recreation, and fish and wildlife habitat. Each of these purposes results in a different type of reservoir operation for storage and discharge and often must be balanced based on water load and water demand. While general operating guidelines have been developed for reservoir operations, interest in optimizing water control for water quality management has recently increased. The objective of this section is to provide a general overview of reservoir distribution, purposes, and operations and to provide a background for enhancement techniques presented later.

1.2.12 DISTRIBUTION AND TYPOLOGY

A dam can be defined as constructed impoundments that are either (1) 25 feet or more in height and greater than 15 acre-feet in capacity, or (2) 6 feet or more in height and greater than 50 acre-feet in capacity (USEPA 1993). Using this definition, there are more than 7,700 dams in the United States. These dams vary in construction and complexity from earthen dams with simple outlet works to large concrete dams with elaborate outlet works. The impounded water upstream from the dam is referred to as a reservoir (pond, lake, and pool are also terms commonly used).

Dams and reservoirs are located to provide water control based on need(s) resulting in a wide distribution but often with logical groupings. These groupings typically include the following types of projects: run-of-the- river, main stem, transitional, or storage reservoir. A run-of-the-river dam has a low dam, limited storage, minimal retention time, low hydraulic head and no positive control over water storage. Discharge is a function of inflow. Mainstem dams are projects located on a major waterway as opposed to a tributary to the river and include run-of-the-river dams. Mainstem dams are characterized by a retention time of about 25 days and a maximum depth between 50 and 100 feet (USEPA 1993). These dams (with the exception of low dams) have water control capabilities and can be utilized for most of the project purposes listed above. Transitional dams display an even greater retention time (up to about 200 days) and have a maximum depth between 100 and 200 feet. The increased retention time and greater depth result in the potential for management opportunities for water control and increased development of water quality patterns. Storage dams are typically more than 100 feet deep and have a retention time greater than 200 days. Maximum hydraulic head at storage dams provides for hydropower generation and storage capabilities provide for maximum flood control. In some areas, dry dams (a dam with no permanent reservoir) are used for flood control. Lock and dam combinations are used primarily on large rivers for navigation and a minimal reservoir or pool is maintained.

While dams and reservoirs are distributed fairly evenly throughout the continental United States, large projects are primarily the responsibility of the U.S. Army Corps of Engineers, the Bureau of Reclamation, and the Tennessee Valley Authority. Each of these agencies have primary responsibilities in various drainage basins. For example, Walker (1981) points out that the CE projects are distributed mostly between 30 and 45^o latitude which is lower in latitude than most natural lakes and the area of glaciation (Figure 1.2.10). The distribution of these projects by type (e.g., reservoir, lock, dry dam) is depicted in Figure 1.2.11.

1.2.13 PROJECT PURPOSES

Most reservoirs are authorized as multiple-purpose projects with storage allocated for two or more purposes. Whether operated separately or as a system (several reservoirs in a river system), demands of the multiple purposes often result in conflicting uses for the reservoir storage. Changes to the water control plan have to be investigated if considerable deviations are required to balance the conflict. The addition of water quality enhancement as a project purpose sometimes contributes to the conflict but can also be used to support a compromise if water quality is enhanced.

Flood control consists of storing excess water (that volume that is above the downstream channel capacity) during flood periods for later release. These later releases may be made during periods of flow below the downstream channel capacity. Since a major factor in flood control reservoirs is maintaining available volume or empty storage space for storage of flood waters, the flood control purpose is generally the least compatible with other project purposes.

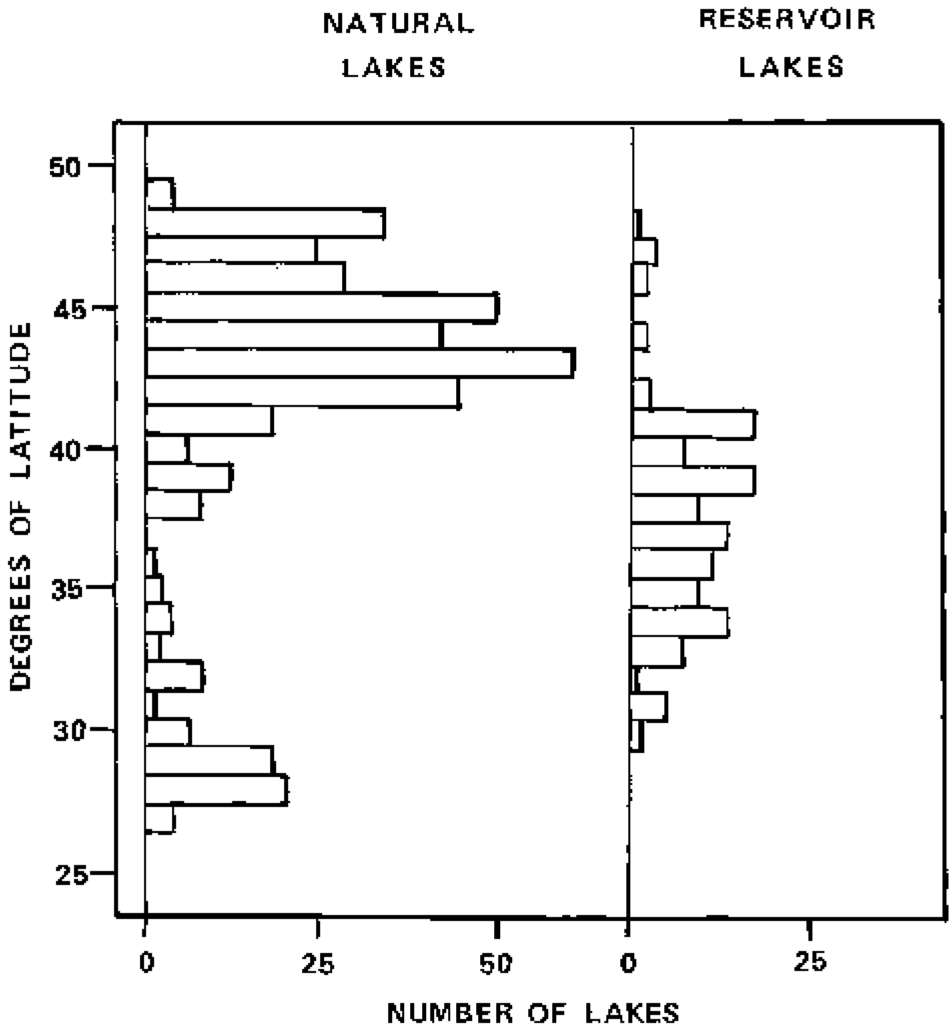


Figure 1.2.10. Distributions of natural lakes and reservoirs
(based on Walker 1981)

Locations of CE Projects

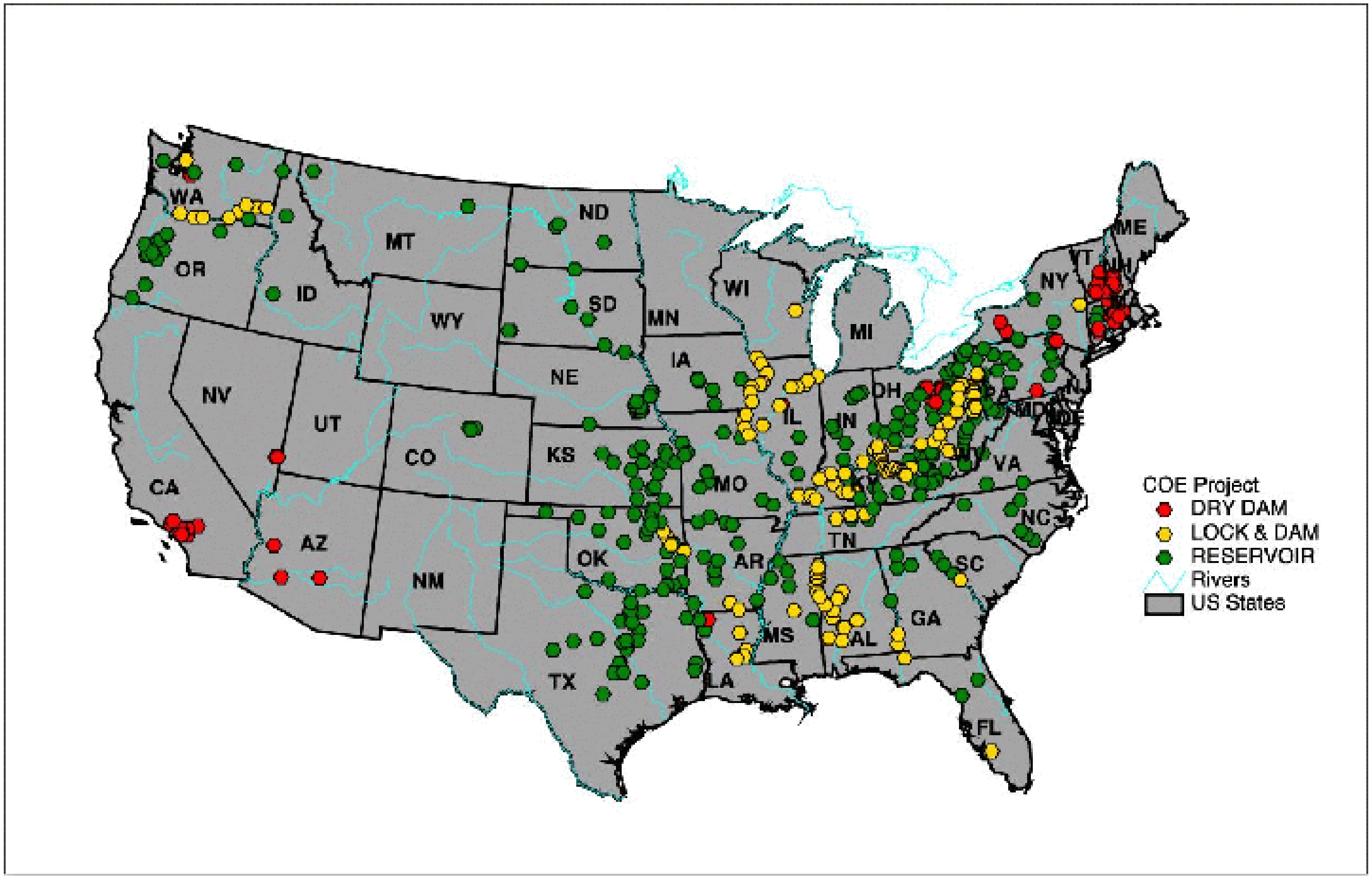


Figure 1.2.11 Distribution of Corps of Engineers projects by type

Hydropower projects are constructed to pass water through turbines to produce two types of power: baseload power and peak power. Baseload power is firm power generated to supply a portion of a constant daily demand for electricity. Peaking power is supplied above the baseload to satisfy variable demands during periods of heavy electricity usage. The power output of a project is determined by the flow through the turbine and the head or pressures exerted on the turbine. Therefore, it is advantageous to have hydropower reservoirs at maximum storage or full storage for maximum hydropower generation. This is best accomplished by pumped-storage reservoirs which maintain a full pool by pumping back into the reservoir following generation. Most hydropower projects are not pumped-storage and are interconnected to a power grid so coordinating generation releases for other uses is possible.

Water supply reservoirs are used to store water during periods of excess inflow for use during other periods. Withdrawal may take place directly from the reservoir, or in downstream reservoir releases. Water is generally provided to municipal, industrial, or agricultural users as reservoir storage rather than a specific volume of water. Consequently, water supply can be obtained by a user from the reservoir as long as there is sufficient water allocated in storage.

Reservoir projects operated for navigation purposes are directed at providing sufficient downstream flow to maintain adequate water depth for navigation and/or providing sufficient water volume for lockages. In many navigation projects, the reservoir pool is part of the channel, so pool levels must be controlled to provide sufficient navigation depths within the pool and downstream area. Downstream releases for navigation purposes may have a distinct seasonal pattern, with higher releases required during the dry season.

Recreation activities in and around reservoir projects include camping, fishing, boating and other water craft related activities, picnicking, swimming and hunting. These activities are related to the quality of the fish and wildlife habitat, water control plan, and terrestrial management. Reservoirs used for fish and wildlife conservation and enhancement may include features such as intake structures to minimize entrapment and entrainment of fish and other aquatic species; outlet and emergency spillway structures to minimize contact of aquatic species with waters supersaturated with dissolved gases and to provide appropriate release water quality. Other structures and release strategies include fish ladders and by-pass systems, maintaining vegetation, and providing for a minimal flow (low-flow augmentation). Low-flow augmentation reservoirs provide releases that increase flow in the downstream channel for downstream fish and wildlife purposes or for downstream water quality control.

1.2.14 RESERVOIR OPERATIONS

Reservoir operations may be grouped by the depth (or layers) from which the released water originates. Typically, these groups are surface, bottom, or mixed releases and release from more than one layer is common. The mechanisms for release from these layers vary widely but a few typical structures are provided for reference (Figures 1.2.12 through 1.2.15).

Surface release can be accomplished with an overflow spillway (usually for uncontrolled releases from floods), Tainter gates (or other types for controlled releases of floods or supplemental spillage), or intake structures with surface withdrawal capabilities (Figure 1.2.12 and Figure 1.2.13). Bottom releases are accomplished with gates conduits near the base of the dam (sluice gates) (Figure 1.2.12) or an intake structure with bottom withdrawal capabilities (Figure 1.2.14). Mixed releases are usually accomplished with an intake structure with ports at several elevations for selective withdrawal (Figure 1.2.14) or are the result of a large withdrawal zone during stratification at hydropower projects with large penstock openings (Figure 1.2.15). Other mechanisms for mixed releases include using combinations of available release features. For example, bottom releases through sluice gates can be augmented with a minimal spill from flood gates to increase aeration. Many hydropower projects also operate a small unit for local power needs and the withdrawal for this unit may not coincide with the withdrawal for the main hydropower units or releases during nongeneration. This type of operation would result in mixed releases. Effects on water quality as a function of release regimes may be evaluated with knowledge of water quality processes as briefly described in preceding sections and can be predicted for selected constituents with models.

Pool fluctuations (changes in surface elevation including the tailwater region) are also determined by reservoir operations and can impact uses such as fish and wildlife habitat and recreation. Generally a water control plan is developed for each project and is defined by project authorization (purposes for which the project was constructed) and an area-capacity curve indicating available water supply at a given elevation (Figure 1.2.16). Operations and water allocation also affect the retention time (defined as the volume divided by the discharge) of the reservoir water which in turn effects the water quality in the reservoir and its release.

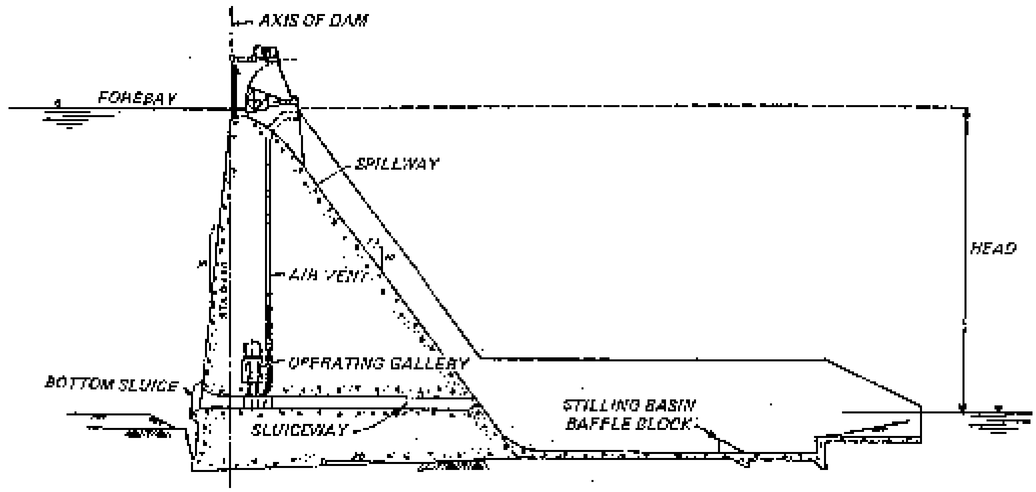


Figure 1.2.12 Illustration of bottom withdrawal structure, spillway,
and stilling basin (USACE 1987)

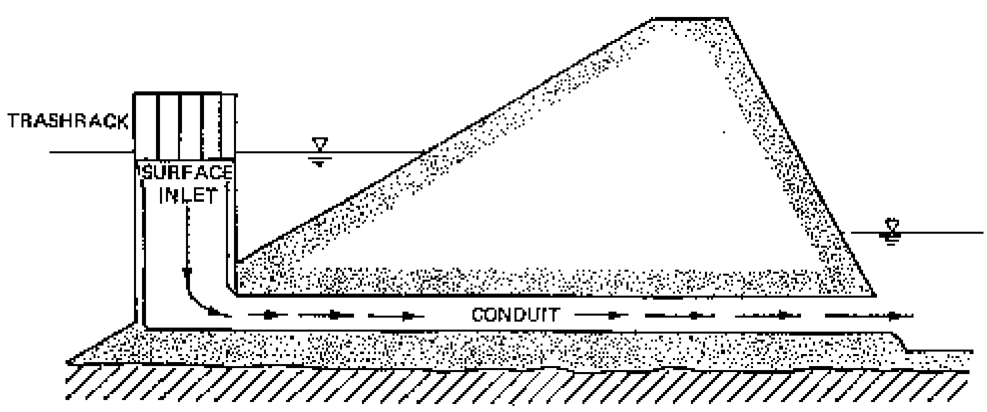


Figure 1.2.13 Example of surface withdrawal structure (USACE 1987)

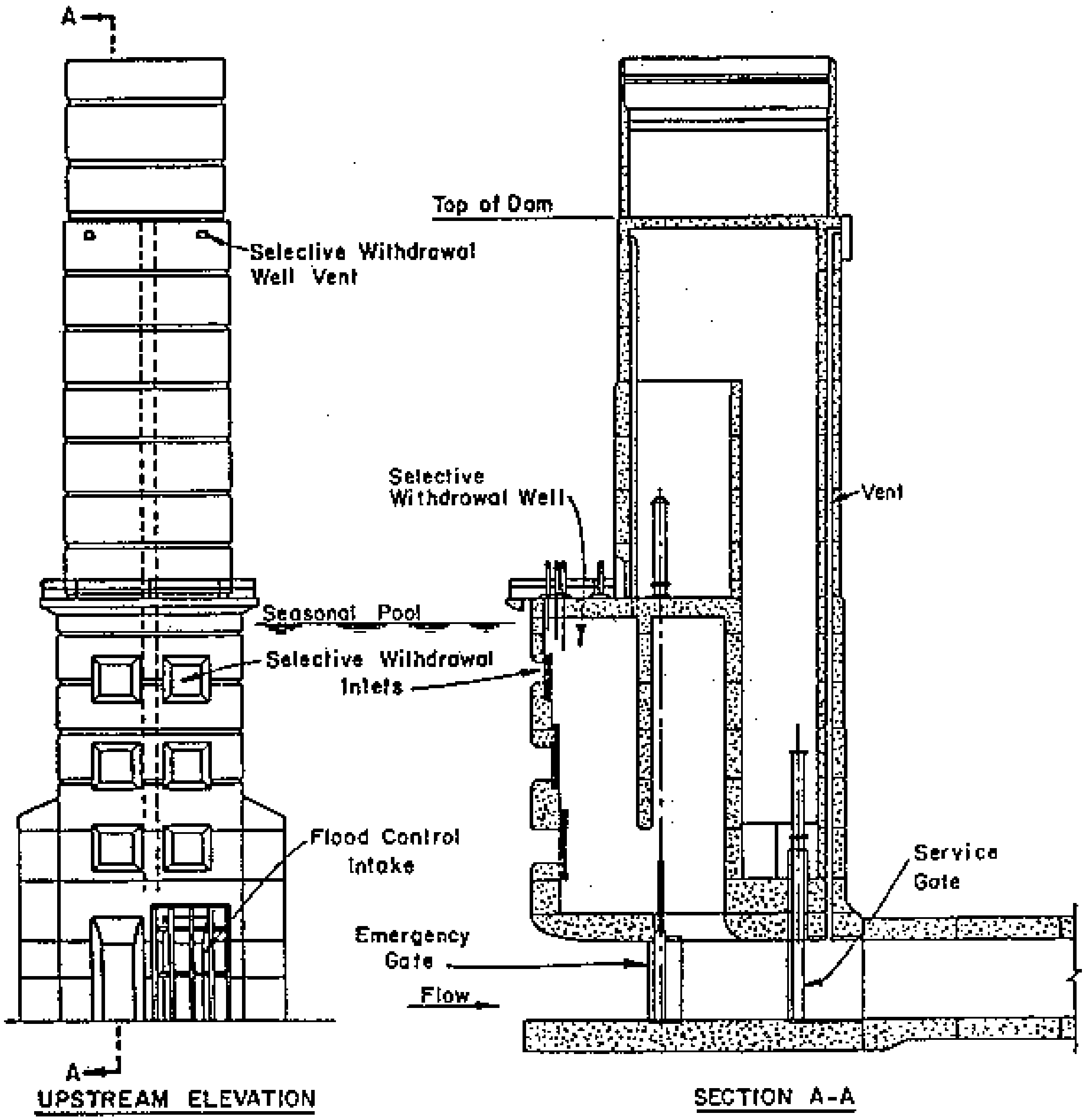


Figure 1.2.14 Dual wet well multilevel withdrawal structure (USACE 1987)

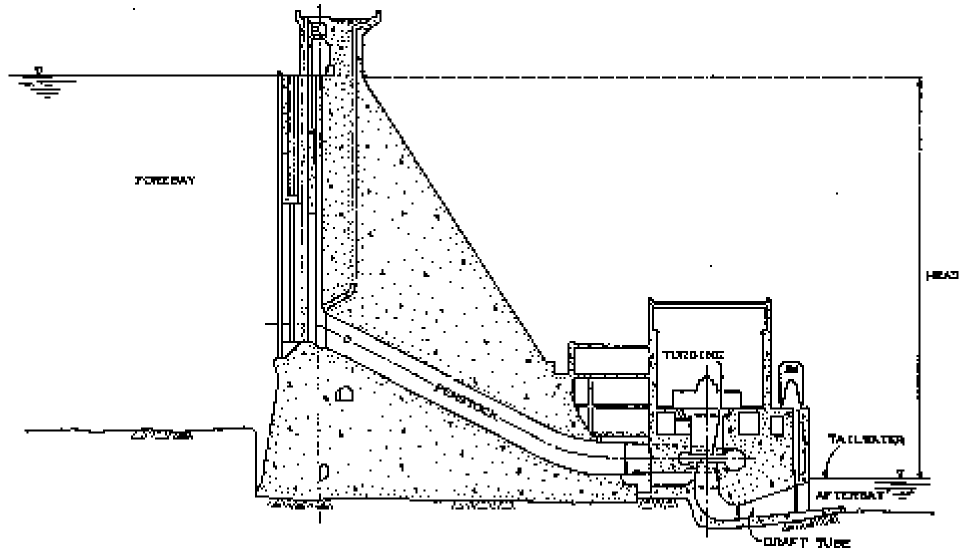


Figure 1.2.15 Schematic of a hydropower facility (USACE 1987)

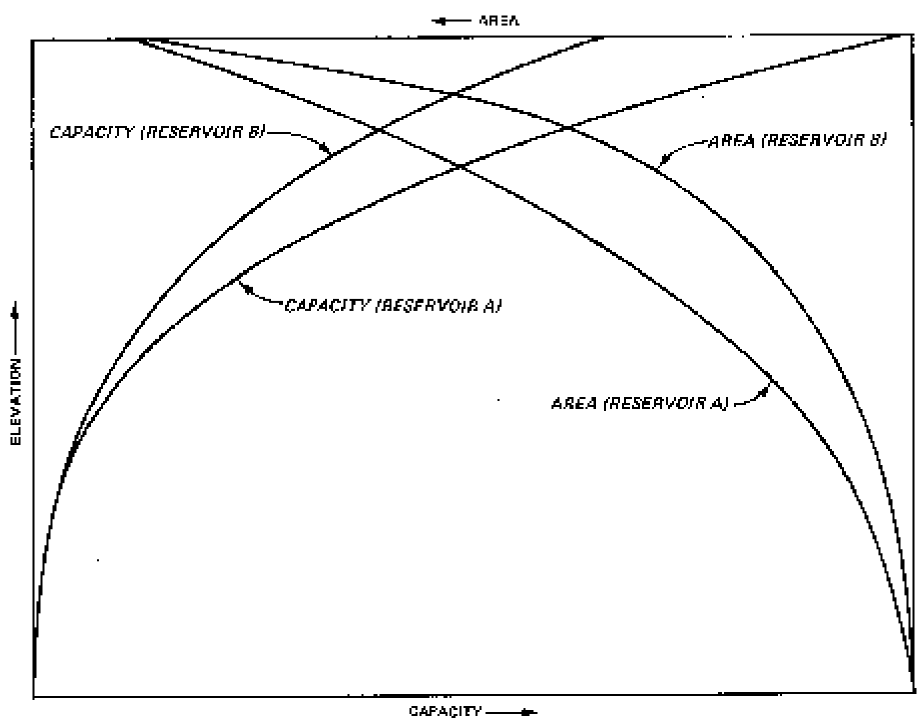


Figure 1.2.16 Area-capacity curves for two reservoirs having similar depth and surface area but differing volume and morphometry

1.2.15 HARMFUL ALGAL BLOOMS AND THE FACTORS CONTROLLING ALGAL GROWTH

1.2.15.1 Introduction

There has been increasing interest in the occurrence of nuisance and harmful algal growth in freshwater environments during the decade ending 2000. This follows a long interest in marine systems related to red tides and shellfish toxicity. Today numerous sites of impairment have been identified in freshwater systems as well as numerous taxonomic groups of harmful algae and numerous toxins that can be produced. The circumstances leading to the excessive growth of algae and plants are understood with some confidence but the factors leading to the growth of specific types of algae are still being investigated. There is very little known about the factors leading to the production of toxins by such algal growths. The purpose of this section is to summarize the body of knowledge associated with this topic as well as the factors contributing to algal growth and potential control measures.

Excessive growth of these plants in an aquatic environment is often termed a "bloom" or an "algal bloom". Some authors have also referred to them as "water blooms". Such blooms are subjectively identified when growth has proceeded to an extent that the presence of the vast numbers of these cells is obvious to unaided vision. However, these blooms can also occur at depths that make them invisible, or at concentrations that do not make them visible but nevertheless have an adverse impact on the resource. Blooms can be harmless, or they can be a nuisance because of esthetics or smell. At their worst, a bloom can eventually consume dissolved oxygen during decomposition or produce powerful toxins dangerous to wildlife and humans. A harmful algal bloom (HAB) is usually associated with adverse effects on water quality including anoxia and toxin production. Anderson and Garrison (1997) provide an extended general description of what constitutes a HAB. "HABs are 'blooms' of microscopic and macroscopic algae that cause harm in a multitude of ways." The entire volume associated with their reference is devoted to HABs and is an excellent place to begin a study of HABs (Limnology and Oceanography 42[5, part 2] 1009-1305).

Most of the contemporary interest in HABs is directed toward certain well-known algal types. The following text is intended to provide a brief acquaintance with the common algae and the specific ones often responsible for HABs. For additional motivation to read the introductory sections, the reader is directed to the later section on the history of algal toxic poisonings.

1.2.15.2 Background

What do the algae include? The algae technically include all single-celled plant organisms. In practice their highly variable growth form separates certain macroscopic algae (the kelps and other giant algae, for example) from the microscopic forms that compose the large number of species in many aquatic environments. The microscopic forms can be found living as single cells such as many diatoms like Navicula spp., green algae such as Cosmarium spp. or

Chlamydomonas spp., the dinoflagellates (Peridinium spp.), and others. Many of the microscopic forms also form multicellular colonies or filaments of single cells. Examples include colonial Volvox spp. and filamentous Oedogonium spp. There are thousands of microscopic algal species.

In freshwater systems, there are also several genera that have cells and colonies large enough to be easily visible. Sometimes these algae are included in manuals on identification and control of aquatic plants. Chara spp., Nitella spp., and Hydrodictyon spp. are examples of such "large" microscopic algae. All three of these belong to the green algae group although there are other large freshwater forms in the red algal group.

In terms of phylogeny and physiology, the green algae are considered to have the most characteristics in common with higher plants. A single spinach cell would have many of the same structural and functional characteristics as, say, a single Chlorella spp. cell.

Major Algal Groups. Although this will be explained later in this section, there is one algal group that is more like bacteria than other algae. The Cyanobacteria are described in a section immediately following this one. All other groups of algae have characteristics in common with higher plants and a cellular structure that includes membrane-bound organelles such as the nucleus, chloroplasts, and mitochondria. This and other characteristics places them among the eukaryotic organisms which include all multicellular plants and animals as well as many other single-celled organisms that are animal-like.

Many biologists place the algae within the eukayotic group that also includes single-celled animals and is called Phylum Protista. Although the algae are technically considered single-celled organisms, as mentioned before, they can form multicelled colonies or multicelled tissues that look very much like higher plants. When the algae form large structures that can be collected by hand or viewed with the naked eye they are sometimes referred to as "macroalgae". Examples of macroalgae include Ulva (sea lettuce), and the many kelps. The difference is that the cells of these algae are still not completely modified into functional tissues such as leaves or vascular tissue. In some cases, single cells taken from these complex organisms can grow into new algal organisms. For a more complete discussion of the differences, the reader should refer to a botany text or a college-level biology text.

The major groups of the algae include some that most people do not normally think about outside of their school courses sometime in the past. The major groups are briefly outlined as follows:

Green Algae - This extremely diverse group includes common filamentous and single-celled organisms such as Spirogyra, Ulothrix, Cosmarium, Chlorella, Staurastrum, Pediastrum, Scenedesmus, Volvox, Chlamydomonas, Pithophora, and Cladophora. This group also includes species that form structures large enough to be easily seen without the aid of a microscope and are considered "macroalgae". The green algae can bloom as thick planktonic suspensions of single cells or colonies, or as thick mats of tangled filaments either attached to substrates or lying on the bottom of the lake. The latter sometimes float to the surface as large filamentous mats and can clog intakes and waterways. The green algae have certain photosynthetic pigments in common and most closely resemble land plants in this and some other respects. A single spinach cell would, in many ways, be quite similar to a single green algal cell. As in all algae these contain Chlorophyll a, the primary photosynthetic pigment. They also contain Chlorophyll b, a common accessory pigment. Their cell walls contain cellulose just like land plants.

Green algae occur in both freshwater and marine environments. Ulva, mentioned as an example before, is a marine alga that forms leaf-like growths resembling lettuce. And as with lettuce, spinach and many green algae, Ulva is edible. (Here I make a distinction between 'edible', an objective judgment, and 'palatable', a subjective judgment.) Two macroalgae commonly found in freshwater are related to each other: Chara (muskgrass) and Nitella. These are common inhabitants of lake sediments in the littoral zone and are large enough to be confused with vascular aquatic plants.

Some green algae float as plankton, dependent on water movements, others are attached to surfaces, and others such as Chlamydomonas can swim quite well with the aid of one or more flagella.

Diatoms and golden-brown algae - This group is sometimes separated into two or more groups consisting of the diatoms and the golden-brown algae. The two groups are similar in their pigment content that also includes Chlorophyll c, another accessory pigment, but they differ in structure. The diatoms are the algae that formed diatomaceous earth, a common filter material and they all form siliceous (glass) structures as their cell walls. These are called 'frustules' and are often shown in microscopic illustrations because of their diverse and beautiful forms. They are common to all waters except the most extreme thermal environments and they are among the most ecologically important organisms on earth. The golden-brown algae also include common planktonic forms such as Dinobryon, Mallomonas, and Synura. These algae do not form the frustules of the diatoms but they often do have siliceous scales. Many of the golden-brown algae can swim with the aid of their flagella. Diatoms, however, either do not move or when attached to substrates can "glide" slowly without flagella. As plankton, the diatoms are considered euplankton, completely subject to water movements.

Euglena-like algae - These algae are motile by means of flagella. The best known genus, Euglena, is typical of the group. Among the well-known characteristics are several animal-like characteristics including the absence of a cell wall (usually found in plants), active motility, and the ability to ingest food particles. Please note that recently other algal forms have been found to be able to ingest food particles as well. This mode of nutrition (mixotrophy) is still being investigated for its importance in aquatic ecosystems. Members of the Euglena group store food as a unique form of starch called paramylon. In contrast with other starches paramylon has the curious property of not changing color in the presence of iodine. Some commonly encountered genera of this group include Euglena, Phacus, and Trachelomonas.

Dinoflagellates - These are very important algae in both freshwater and marine ecosystems. They possess flagellar motility and are capable of active migration in the water column. During times of stratification, they can exert diel changes of position, presumably in association with light intensity or nutrients. Examples in freshwater include Peridinium and Ceratium. In marine systems blooms of these form the famous 'red tides' that occur with increasing frequency and intensity in recent times. The most common 'red tide'-forming genus is Gymnodinium. In marine systems these are well-known for their ability to form powerful toxins which when concentrated in shellfish can cause paralytic shellfish poisoning (PSP). There will be more discussion of this later in this section. Freshwater species are seasonally abundant and are an important component of the aquatic ecosystem. These are also capable of both photosynthetic food production as well as ingestion of food particles although the mechanism of ingestion does not involve highly organized structures as in Euglena.

Red algae - These have very complex life cycles, unique pigments and storage products, and varied micro- and macroalgal growth forms. Red algae are never found in the plankton, and are seldom, if ever, found in lakes and reservoirs. In freshwaters, the best-known of these is Batrachospermum, found in freshwater streams in the spring. Marine systems have numerous examples of red algae, some of which are used as a food source by humans. At this time these are not among the HAB forming organisms of freshwaters.

Other groups - There are other unique but inconspicuous algal types in freshwaters. For example a single species, Gonyostomum semen, is commonly found in stratified, productive lakes and reservoirs. Although it may be common, perhaps dominant, in a microenvironment near the thermocline, it is seldom collected and accounted for because of its fragility. Lacking a cell wall and possessing trichocysts, these cells tend to be destroyed by normal collection and preservation methods. Other unique algal species, like Gonyostomum, await further research to account for their contribution to the aquatic ecosystem.

Cyanobacteria: the Blue-Green Algae. One group of algae that commonly contribute to nuisance growths is technically considered not to be algae by many scientists. The so-called "blue-green algae" are actually prokaryotic organisms whereas the species named and described in the previous paragraphs are eukaryotic organisms. The distinctions between these two types of organisms are many and they are important to the environmental effect they can have on aquatic systems. As stated earlier the prokaryotic "blue-green algae" do not have cellular organelles (membrane bound structures like the nucleus, chloroplast, or mitochondrion). The "blue-green algae" actually have more features in common with bacteria, which also are prokaryotic organisms. For this reason the "blue-green algae" are correctly referred to as Cyanobacteria. This section will refer to these organisms as cyanobacteria although it is still acceptable to refer to them as "blue-green algae" and there are still numerous references to that terminology in the literature. The cyanobacteria include many taxa that have a variety of growth forms. They can occur as single small cells or in aggregations of small single cells. They also grow in filaments (or similar structures called trichomes) but they are usually microscopic. The large aggregations of these forms can easily be seen without the aid of a microscope such as scums of Microcystis and Anabaena, the suspended clumps of Aphanizomenon, and large mats of Oscillatoria or Lyngbya. There are three genera that historically have caused nuisance blooms in lakes. These three have been referred to humorously as "Annie, Fannie, and Mike". Their actual names are Anabaena spp. (Annie), Aphanizomenon spp. (Fannie), and Microcystis spp (Mike). What they and other species are capable of is not humorous, however, and there exists a perception that problems with blooms of these and other taxa are increasing.

Taken together, the cyanobacteria and the algae compose a highly diverse assemblage of organisms that can do many things in aquatic environments. They can grow on surfaces, among sand grains, float on the surface as a scum, accumulate in filamentous masses on the bottom (floating up at any time), and some can control their position through active motility (swimming) or changes in buoyancy. They can utilize nutrients in different forms and some of the cyanobacteria can actually add nutrients to the aquatic system. Most photosynthesize, making their own food (autotrophy) but some can also consume food particles (mixotrophy) and can exist for times exclusively on external sources of food (heterotrophically).

And importantly, some have been identified as the source of very potent toxins in aquatic ecosystems. This capability has been known for decades but the importance of this characteristic has become widely known only during the last 10 or so years.

Other Important Differences Between Cyanobacteria and True Algae. The single most important difference between cyanobacteria and algae has been described above. However, there are other differences that are also important. Some of these are listed here:

1. The cyanobacteria contain additional photosynthetic pigments, phycoerythrin and phycocyanin, that allow use of light unavailable to other algae.
2. The cyanobacteria are capable of utilizing free gaseous nitrogen as a nutrient source. This is called 'nitrogen fixation' and only occurs in these organisms and in the bacteria (remember root nodules in members of the pea family?).
3. The cyanobacteria are unable to swim as in some green algae. However, they can alter their buoyancy allowing them to control their depth in the water.
4. The cyanobacteria can efficiently store nutrients during luxuriant growth and uptake. Then these can be utilized for nearly 10 additional doublings of cell growth under subsequent low nutrient conditions. Algae have less ability for such storage.
5. Cyanobacteria can produce different toxins and noxious substances from algae. However, some of the toxins may be similar to those found in red tides.

1.2.15.3 Brief History of Toxic Algal Poisonings

Chorus and Bartram's (1999) book gives a brief history of cyanobacterial toxic poisonings in the introduction. Because the book is limited to toxic cyanobacteria, it is important to note that if accounts of poisonings due to other algae such as dinoflagellates had been included, the history would be much richer but probably not older.

Francis (1878) is reported to be the earliest documentation of cyanobacterial poisoning (of livestock in that instance). However, earlier anecdotal accounts extend to nearly 1000 years earlier to the Han Dynasty in China when a general reported poisonings of troops that drank from a river that was green in color. Codd (1996) stated that awareness of toxic blooms existed in the 12th century at the former Monasterium Virdis Stagni near Soulseat Loch in Scotland. This anecdotal awareness has become manifest in the practice on several continents where people dig shallow depressions near waters containing green scums in order to 'filter' out the scum for drinking purposes. More recently there have been some alarming incidents involving both cyanobacteria and dinoflagellates.

In 1989, 16 British Army recruits training on a Staffordshire reservoir became ill with pneumonia. Two of the recruits received extensive treatment for one week. This was eventually attributed to exposure to water containing a harmful algal bloom (HAB) (Turner et al. 1990). A record of human poisoning from drinking water in Queensland, Australia extends back to 1887 and has recently been attributed to toxic cyanobacteria in those inland waters (Hayman 1992). Algal toxins from Malpas Dam in New South Wales, Australia affected the human population in 1973 (Falconer, Beresford, and Runnegar 1983). Similar observations regarding liver cancer have been made in China (Ueno et al. 1996). Sixty haemodialysis patients died from algal toxins contained in the dialysis water derived from the municipal water supply for Caruaru, Brazil, and 66 additional patients developed acute toxicity symptoms (Pouria et al. 1998). Pets, waterfowl, livestock, and wildlife are affected in both natural lakes and reservoirs in this country. Coastal waters associated with or near Corps navigation projects increasingly experience "red tides" and blooms of algae as toxic as those described above. The well-documented occurrence of Pfiesteria in estuarine and near-shore waters has been cause of alarm and health impairment for nearly a decade.

Four major types of poisoning have been described from toxins produced by **marine** HABs. These are: paralytic shellfish poisoning (PSP), neurotoxic shellfish poisoning (NSP), diarrhetic shellfish poisoning (DSP), and amnesic shellfish poisoning (ASP). As the names imply they have a common mode of human impact. These marine toxins are all produced by dinoflagellates except ASP, which is produced by a marine diatom (Plumley 1997). This was of sufficient interest to hold the First International Conference on Toxic Dinoflagellate Blooms in 1974 (LoCicero 1975). In contrast, toxins in freshwaters are produced primarily by cyanobacteria and they are apparently more diverse than marine toxins (also Plumley 1997).

Numerous Internet sites exist covering the topic of harmful algal blooms…just search "harmful algal bloom" or "HAB". Similarly, searching "Pfiesteria" will also identify numerous sites and bibliographical information resources on this toxic dinoflagellate.

1.2.15.4 Nuisance Growth and Harmful Algal Blooms - Growth and Control

What is a Nuisance Growth? A nuisance growth is defined by the user of the resource in which the growth occurs. In the same manner that a growth on a person could be viewed as a distinguishing mark (a freckle, for example) often such things are viewed as a nuisance. If they pose a medical threat, then such growths are considered health risks. Similarly, in the aquatic environment, a small quantity of aquatic plant growth may be viewed as interesting, beneficial, or even sightly. When that growth chokes the waterway it is a nuisance, possibly harmful. When it produces toxins, it is a health risk. When there is resultant economic or biological damage, the growth is a harmful algal bloom (HAB).

What Causes Nuisance Growth? Plants grow or die. There is no compromise. Aquatic systems constantly change with flows, seasons, resource use, and numerous other factors. The interactions among all these factors have led to some generalizations on what contributes to plant growth.

Plants and algae require water, light, and nutrients. While temperature obviously affects the growth of all organisms, there is usually some species that are well adapted to all normally occurring aquatic temperatures. In aquatic environments the only two factors that usually limit algal growth are light and nutrients.

>>TRUTH STATEMENT<<: In nature, as long as a habitat offers sufficient or abundant resources to support growth, **something** will attempt to utilize those resources and grow abundantly.

Because the growth controlling resources are often keys to long-term management, understanding how they affect plant growth is therefore also important. The two types of resources necessary to support plant growth are briefly described below. They include the source of energy for growth and the materials needed to support that growth.

Light Energy. The sun is the only source of the light energy that is necessary for photosynthesis and plant growth. The same factors controlling the distribution of light energy in water and its absorption by water affect the availability to plants in the water. There are obviously depths in deeper lakes to which no light effectively penetrates and plants do not exhibit photosynthetic activity there. Also, because longer wavelengths are usually absorbed nearer the surface, both the light quality and intensity changes with depth. For this reason, at certain depths of many lakes, there are specific types of algae adapted to use that light and to live at that range of depths. The factors controlling available light have been addressed elsewhere in this course.

Nutrients and Eutrophication. Cultural eutrophication began in North America with the arrival of European settlers. Cultural eutrophication differs from the original concept of eutrophication by virtue of its speed and the processes contributing to it. The original concept of eutrophication applied to the natural tendency of a lake to change with age.

In the original concept, the young natural lake, formed through some natural process like tectonic activity or glaciation, was initially at its deepest and soon was surrounded by well-forested and protected watershed. Over great time sediments accumulated and the lake depth decreased. As the lake became shallower, plants begin to grow along the littoral zone where light could penetrate to the shallow sediments. This then tended to allow materials to accumulate more efficiently in those areas and the sediments become richer in nutrients. With a great amount of additional time the lake became shallow enough to be considered a wetland with aquatic plants emerging in the middle of the old lake bed. Finally trees and other plants colonized the drier sediments. This 'death' or "eutrophication" of a natural lake was later applied to the rapid processes associated with the artificial nutrient pollution during recent times that contributed to some of these aging processes. Man-induced nutrient enrichment is considered "cultural eutrophication" because it is largely due to the characteristics of our culture that we contribute to those nutrient additions.

Earlier in the 20th century there were developed a number of indices for comparing the status of different lakes and other surface waters. The most recent example of such an index is the trophic state index or TSI (. This is based on the relationship between a number of related lake characteristics such as water clarity (secchi depth), nutrient concentration, chlorophyll concentration, and others. This index is a useful tool for initial comparisons but its applicability must be carefully judged for each situation.

In reservoirs, eutrophication is usually viewed in terms of nutrient enrichment and the outcome of nutrient enrichment because there is no historical perspective for comparison. Most reservoirs (nearly all the large ones) were built during the 20th century and despite that, some have already substantially filled with sediments. This is an outcome of some of the differences between natural lakes and reservoirs that were outlined earlier in this course.

In northern natural lakes, cores of old accumulated sediments clearly show the shift of terrestrial plant pollen types from forests dominated by oaks and other trees to grasses and the definitive early successional plant, Ambrosia (ragweed), that became common after the forests were cut by the first settlers. This pollen record not only tells the story of ecosystem change in the landscape, it also parallels the change in sedimentation rate, and the changes in the types of organisms living in the lake. Algal fossils (diatoms, mostly) tell of sudden changes of species associated with cultural eutrophication the result of increased nutrient and sediment loading to the aquatic environment. Similar trends have also been observed in the patterns of zooplankton fossils in old lake sediments. These changes have continued for hundreds of years in North America, thousands of years in Europe and Asia, and in the 1950's and 60's the changes accelerated rapidly with increased use of phosphate detergents, pesticides, development, agriculture, and other impacts to the watershed.

Ecological Growth Control. Nuisance growth of algae (and other plants) is related to eutrophication (cultural eutrophication), nutrient loading, sedimentation, land use, and many other factors. The specific causes of nuisance growth relate to most of those factors. Ecologists are divided in their assessment of the factors controlling algal growth. While it is acknowledged that all of the factors can affect growth, these factors are divided into three categories related to trophic relationships in the ecosystem. "Bottom up" controls include those influencing the supply of basic requirements for growth such as nutrients or light. In this view the algal species compete for nutrients or available light and this competitive interaction coupled with specific growth potential determines the amount and types of algal species. The term "bottom up" implies that the base of the food web controls overlying or dependent components of the food web. Another view is that food web control is "top down". In this view, the presence or absence of different predators will control the types of herbivores. The types of herbivores then influence herbivory which in turn influences the amount and types of algae dominating the ecosystem. "Top down" implies that higher food web components (such as fish) control the growth and types of the food items they depend upon.

The other factors such as temperature, climate, geology, etc. fall into the category of independent factors not controlled by biological processes. All of the above factors are listed below:

1. Nuisance growth cannot occur where conditions cannot support luxuriant plant growth. Therefore, the essential elements for growth must be present in excess. The limiting nutrient must be found in sufficient concentration. Phosphorus is commonly implicated as the limiting nutrient and is one of the most common components of pollution. Any essential nutrient can theoretically be a limiting nutrient. Others commonly implicated are N, C, Si, and Fe (mostly marine).
2. Light must be available. Excessive turbidity or other mechanisms blocking the light will tend to limit growth even where nutrients are in excess. Seasonal trends are also important in as much as some latitudes have longer summer days with less intense light while other latitudes have shorter days with more intense light.
3. Herbivory by zooplankton or other animals can limit the standing crop of plant biomass. This is the “sledgehammer” approach to plant control if, for example, a herbivore such as the grass carp is introduced to remove aquatic plants. In the plankton, a prolific population of planktivores in a good year can shift the dominant phytoplankton species.
4. Herbivory can also be selective, allowing less-grazed species a selective advantage.
5. Changes in predators on herbivores can change patterns of herbivory. This may affect the selective pressure on the plant species, causing a shift in species dominance. Although well-described, this is often ignored in such management decisions as creel or size limits for fish populations. If ignored, the 'bottom up' control assumption is implied.
6. Physical processes affecting the distributions and movements of supporting resources are also important. The most important of these is water movements, either flows or turbulence.

Factors Under Hydro-project Control. This course presents a number of means of improving or managing water quality in reservoirs and tailwaters. When modifications to project operation are made for water quality considerations these are usually related to the temperature or dissolved oxygen concentrations in releases and, more rarely, in the lakes. However, it is important to remember that anything that alters the distribution of dissolved oxygen in the hypolimnion can also alter the nutrient distributions in a reservoir. And any change that alters the movement of water through a reservoir will also affect material distributions including nutrients and organisms. Project modifications can alter the distributions and types of biota living in the reservoir.

Nutrient Ratios: When a best management practice (BMP) is successful it may have a perceived positive influence on water quality in downstream systems. This perception is sometimes arguable. If a lake is negatively impacted by nutrient loading (eutrophication) the most common nutrient that is implicated is phosphorus (P). Reduction of P loading is often a major goal of BMP's for use or development in a watershed. Sometimes reservoirs can accomplish this in complex river systems. For example, Richard B. Russell Lake (RBR) greatly reduced the amount of P entering J. Strom Thurmond Lake via the Savannah River. In this sense RBR acts as a nutrient trap.

If an ecosystem is adapted to certain available nutrient concentrations then changing one of them will change the ratio of available nutrients. This can affect the limiting status of the essential elements.

The simplest application of this idea applies logic developed around the "Redfield ratio" (Redfield 1934 and Pearsall 1930, 1932). This ratio relates carbon, nitrogen, and phosphorus and is approximately 120:16:1, respectively. In practice the ratio is often expressed simply as nitrogen (N) to phosphorus (P) ratio because it is rare for carbon to be more limited than either N or P in aquatic systems (although it does happen from time to time). Aquatic ecosystems can be either N- or P-limited but P is the most common limiting nutrient. In extremely oligotrophic lakes, both N and P can be limiting and in extremely eutrophic lakes, light (rather than nutrients) can become limiting.

This is important to the algae because changing nutrient status may change the dominant form of algae in the aquatic system (Kilham and Kilham 1984). For example, reduction of N-loading without similar reductions in P-loading can result in N-limitation. This can give algae that are capable of N-fixation a competitive advantage over those that can't and may shift the system from green algae, for example, to cyanobacteria (Smith 1983). Although the biomass may not change or it may even decrease, other qualitative aspects of this changed system may be judged not to be improvements.

Hydraulics: One means of improving water quality that is being increasingly used in reservoirs is in-lake aeration or oxygenation. This topic is presented in great detail during other parts of this course but one aspect of it is gemaine to HABs. If an aeration or oxygenation system is operated in such a manner as to mix a portion of the reservoir, then this water movement (mixing or turbulence) can greatly alter the growth potential or the types of dominant organisms in the lake.

Water movement (flow) has long been known to stimulate algal growth. Lawrence Whitford (1960) noted that Ruttner (1926) had first begun to describe this effect. Whitford quoted Ruttner (1953), "In quiet or in weakly agitated water the organisms are surrounded by a closely adhering film of liquid, which speedily forms around the animal or plant, a cloak impoverished of substances important for life. In a rapid current, however, the formation of such exchange-hindering investitures is prevented, and the absorbing surfaces are continually brought into contact with new portions of water as yet unutilized." Ruttner continued to note that moving water "is not absolutely but rather physiologically richer in oxygen and nutrients."

Under some conditions the type of movement and its intensity can produce different results. The intensity and duration of turbulent mixing affects both the total production as well as the type of algal dominant. Sufficient turbulence can also disrupt structural integrity of cyanobacterial filaments and colonies (Thomas and Gibson 1990a & b). Whereas nutrient limited conditions can be overcome through low-intensity turbulence (Rhee et al. 1981) and thus promote growth, increased turbulence can inhibit growth (Paerl 1990).

To provide a mitigating effect to water quality, mixing must be of sufficient intensity to overcome the self-positioning tendency of cyanobacteria that produce gas vesicles (Reynolds 1987). Artificial mixing must continue throughout the favorable growth period for cyanobacteria if it is to be effective in limiting growth (Smith 1988, Shapiro 1990). Even short quiescent periods are sufficient to allow rapid re-emergence of cyanobacterial blooms (Paerl 1987). The decision to address algal growth problems using hydraulic mixing is a complex decision that requires a firm commitment to the treatment.

<u>Nutrient Limitation</u>: Lakes with existing algal blooms can be enhanced by controlling the supply of nutrients to the aquatic ecosystem. However, such nutrient control may be less feasible in certain lakes unless costly measures are taken to accomplish the control. The primary nutrients of concern are N and P. Control of P has been effected in many aquatic environments.

Although decreasing P loading to lakes has been the common goal of many attempts to control eutrophication, our understanding of the ecological processes controlling the distribution and abundance of specific algal types is incomplete. Only the most general statements can be made and any of them may be violated in specific lakes under specific circumstances.

A lake dominated by a nuisance growth of macrophytes and green algae may respond favorably to decreased P loading. However, if sediments contain large quantities of P, then they can potentially become a source of P to the lake rather than the sink they were during sediment P accumulation. A long history of eutrophication may require a lengthy or expensive time for recovery from eutrophication.

If nitrogen controls are engaged, then N limitation in the presence of abundant P can occur. This has the potential for shifting the competitive advantage from eukaryotic algae to cyanobacteria capable of N-fixation (Smith 1983, Howarth et al. 1988). Cyanobacteria are slow-growing compared to eukaryotic groups such as green algae. But once loaded with nutrients, they can grow in the absence of external nutrients sources and can increase their biomass from 4 – 32 times without further nutrient inputs. This opportunistic capability makes them very successful competitors among the algae.

Control of biota in a lake can mean many things. When control is attained using chemical means, decomposition of the eradicated organisms can demand oxygen and release their nutrient content back to the lake. The elemental nutrient mass has been converted from a utilized, bound form (biota) to a potentially available form. Because of these and other sources of uncertainty, management of HABs remains unpredictable in outcome.

1.2.15.5 Future Outlook for HABs

Harmful algal blooms will continue in many lakes and aquatic systems where they already exist. They will become more intense where long-term water quality trends allow them to. At present there is no good predictive tool for management or planning for aquatic resource utilization. Indeed, there is no good comprehensive survey for their distribution or prevalence in the aquatic ecosystem. Further study of their effect has identified additional toxins produced by the cyanobacteria and this study has accompanied the development of analytical tools for identification of the algal strains responsible for certain toxins as well as for the toxins themselves. There has been an increasing awareness of the importance of these organisms in our aquatic resources.

1.2.15.6 Specific References

Anderson, D.M. and D.L. Garrison. 1997. Preface. Limnology and Oceanography 42 (5, part 2).

Chorus, I. and J. Bartram. 1999. Toxic cyanobacteria in water. A guide to their public health consequences, monitoring and management. E. & FN Spon. London. 416 pp.

Codd, G.A. 1996. Harmful Algae News. IOC of UNESCO. 15:4.

Falconer, I.R., A.M. Beresford, and M.T.C. Runnegar. 1983. Evidence of liver damage by toxin from a bloom of the blue-green alga, Microcystis aeruginosa. Med. J. Australia. 1:511.

Hayman, J. 1992. Beyond the Barcoo – probable human tropical cyanobacterial poisoning in outback Australia. Med. J. Australia. 157:794.

Howarth, R.W., R. Marino, and J.J. Lane & Cole. 1988. Nitrogen fixation in freshwater, estuarine and marine ecosystems. 1. Rates and importance. Limnology and Oceanography 33: 619-687.

LoCicero, V.R. 1975. (ed.) Proceedings of the first international conference on toxic dinoflagellate blooms. The Massachusetts Science and Technology Foundation, Wakefield, MA. 541 pp.

Paerl, H.W. 1987. Dynamics of blue-green algal blooms in the lower Neuse River, NC: causative factors and potential controls. Univ. of N.C. Water Resources Research Inst. Report No. 229, Raleigh, NC. 164 pp.

Paerl, H.W. 1990. Physiological ecology and regulation of N2 fixation in natural waters. Advances in Microbial Ecology. 11:305-344.

Plumley, F.G. 1997. Marine algal toxins: Biochemistry, genetics, and molecular biology. Limnology and Oceanography 42(5, part 2): 1252-1264.

Pouria, S., A. de Andrade, J. Barbosa, R.L. Cavalcanti, V.T.S. Barreto, C.J. Ward, W. Preiser, G.K. Poon, G.H. Neild, and G.A. Codd. 1998. Fatal microcystin intoxication in haemodialysis unit in Caruaru, Brazil. The Lancet. 352:21.

Reynolds, C.S. 1987. Cyanobacterial water blooms. Advances in Botanical Research. 13:67-143.

Rhee, G.Y., I.J. Gotham, and S.W. Chisholm. 1981. Use of cyclostat cultures to study phytoplankton ecology. In: Continuous Culture of Cells. Vol.2, 208 (Ed. by P.C. Calcott), pp. 159-186. CRC Press, Cleveland, OH.

Ruttner. F. 1926. Bermerkungen uber den Sauerstoffgehalt der Gewasser und dessen respiratorischen. Wert. Naturwissenschaften. 14:1237-1239.

Ruttner, F. 1953. Fundamentals of Limnology. Univ. of Toronto Press. 307 pp.

Shapiro, J. 1990. Current beliefs regarding dominance of blue greens: the case for importance of CO2 and pH. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie. 24:38-54.

Smith, D.W. 1988. Phytoplankton and catfish culture: a review. Aquaculture. 74:167-189.

Smith, V.H. 1983. Low nitrogen to phosphorus ratios favor dominance by blue-green algae in lake phytoplankton. Science 221:669-671.

Thomas, W.H. and C.H. Gibson. 1990a. Effects of small-scale turbulence on microalgae. Journal of Applied Phycology. 2:71-77.

Thomas, W.H. and C.H. Gibson. 1990b. Quantified small-scale turbulence inhibits a red tide dinoflagellate Gonyaulax polyedra Stein. Deep Sea Research 37:1583-1593.

Turner, P.C., A.J. Gammie, L.T. Hollinrake, and G.A. Codd. 1990. Pneumonia associated with cyanobacteria. Br. Med. J. 300:1440.

Ueno, Y., S. Nagata, T. Tsutsumi, A. Hasegawa, M.F. Watanabe, H-D Park, G-C Chen, G. Chen, and S-Z Yu. 1996. Detection of microcystins, a blue-green algal hepatotoxin, in drinking water sampled in Haimen and Fusui, endemic areas of primary liver cancer in China, by highly sensitive immunoassay. Carcinogenesis. 17(6):1317.

Whitford, L.A. 1960. Ecological distribution of fresh-water algae. pp. 2-10. In: C.A. Tryon, Jr. and R.T. Hartman (eds.) The Ecology of Algae. Special Publication #2, Pymatuning Laboratory of Field Biology, Univ. of Pittsburg, PA.

1.2.15.7 General References

Steffensen D.A. et al. 1994. Neurotoxins from Australian Anabaena IN: GA Codd, TM Jefferies, CW Keevil, and E. Potter (eds). Detection methods for Cyanobacterial Toxins, pp 45-50.

Rapala, J., et al. 1993. Anatoxin-a concentration in Anabaena and Aphanizomenon under different environmental conditions and comparison of growth by toxic and non-toxic Anabaena strains. J Appl. Phycol. 5:581-591.

Harada K., et al. 1991. Microcystins from Anabaena flos-aquae NRC 5525-17. Chem. Res. Toxicol. 4:535-540.

Carmichael, W. 1986. Algal Toxins. Adv. Bot. Res. 12:47-101.

Negri AP and Jones GJ. 1995. Bioaccumulation of paralytic shellfish poisoning (PSP) toxins from the cyanobacterium Anabaena circinalis by the freshwater mussel Alathyria condola. Toxicon. 33, 667-678

Negri AP, Jones GJ and Hindmarsh M. 1995. Sheep mortality associated with paralytic shellfish poisons from the cyanobacterium Anabaena circinalis. Toxicon 33, 1321-1329

Negri AP, Jones GJ, Blackburn SI, Oshima Y. and Onodera. 1996. Effect of culture and bloom development and sample storage on paralytic shellfish posions in the cyanobacterium Anabaena circinalis: J. Phycology (in press)

Jones GJ and Negri AP. 1996. Persistence and degradation of cyanobacterial paralytic shellfish poisons (PSPs) in freshwaters. Water Research (in press).

Jones GJ and Negri AP. 1995. Blooming Algal neurotoxins. Chemistry in Australia, June, 12-13

Jones GJ and Korth W. 1995. In situ production of volatile odour compounds by river and reservoir phytoplankton populations in Australia. Water Sci. Tech. 31, 145-151

Bowmer KH et al. 1992. Physiology of geosmin production by Anabaena circinalis isloated from the Murrumbidgee River, Australia. Wat. Sci, Technol. 25, 259-267

1.2.15.8 Pfiesteria References

Steidinger, K.A., J.M. Burkholder, H.B. Glasgow Jr., C.W. Hobbs, E. Truby, J. Garrett, E.J. Noga & S.A. Smith (1996) Pfiesteria piscicida gen. et sp. nov. (Pfiesteriaceae, fam. nov.), a new toxic dinoflagellate genus and species with a complex life cycle and behavior. J. Phycol. 32:157-164, and cover.

Burkholder, J.M. & H.B. Glasgow Jr. (1995) Interactions of a toxic estuarine dinoflagellate with microbial predators and prey. Archiv. Protistenk. 145:177-188.

Burkholder, J.M., H.B. Glasgow Jr. & K.A. Steidinger (1995) Stage transformations in the complex life cycle of an ichthyotoxic "ambush-predator" dinoflagellate, pp. 567-572. In: Harmful Marine Algal Blooms, by P. Lassus, G. Arzul, E. Erard, P. Gentien & C. Marcaillou (eds.). Paris, Technique et Documentation - Lavoisier, Intercept Ltd.

Burkholder, J.M., C.W. Hobbs & H.B. Glasgow Jr. (1995) Distribution and environmental conditions for fish kills linked to a toxic ambush-predator dinoflagellate. Mar. Ecol. Prog. Ser. 124:43 -61.

Glasgow, H.B. Jr., J.M. Burkholder, D.E. Schmechel, P.A. Tester & P.A. Rublee (1995) Insidious effects of a toxic dinoflagellate on fish survival and human health J Toxicol. Environ. Health 46:501-522.

Steidinger, K.A., E.W. Truby, J.K. Garrett & J.M. Burkholder (1995) The morphology and cytology of a newly discovered toxic dinoflagellate, pp. 83-88. In: Harmful Marine Algal Blooms, by P. Lassus, G. Arzul, E. Erard, P. Gentien & C. Marcaillou (eds.). Paris, Technique et Documentation - Lavosier, Intercept Ltd.

Lewitus, A.J., R.V. Jesien, T.M. Kana, J.M. Burkholder, H.B. Glasgow Jr. & E.May (1995) Discovery of the "phantom" dinoflagellate in Chesapeake Bay. Estuaries 18:373-378.

Burkholder, J.M., H.B. Glasgow, E.J. Noga & C.W. Hobbs (1993) The Role of a New Toxic Dinoflagellate in Finfish and Shellfish Kills in the Neuse and Pamlico Estuaries. Report No. 93-08, Albemarle-Pamlico Estuarine Study. Raleigh, North Carolina Department of Environment, Health & Natural Resources and U.S. Environmental Protection Agency - National Estuary Program, 58 pp. (3rd printing).

Noga, E.J., S.A. Smith, J.M. Burkholder, C.W. Hobbs & R.A. Bullis (1993) A new ichthyotoxic dinoflagellate: Cause of acute mortality in aquarium fishes. Vet. Rec. 133:48-49.

Burkholder, J.M., E.J. Noga, C.W. Hobbs, H.B. Glasgow Jr. & S.A. Smith (1992) New "phantom" dinoflagellate is the causative agent of major estuarine fish kills. Nature 358:407-410, Nature 360:768.

1.2.15.9 Domoic Acid And *Pseudo-Nitzschia* References

General and Review References

Addison, R.F. and J.E. Stewart. 1989. Domoic acid and the eastern Canadian molluscan shellfish industry. Aquaculture 77: 263-269.

Annon. 1992. Domoic acid intoxication. Canadian Commun. Dis. Rep. 18: 118-120.

Bates, S.S. 1997. Toxic phytoplankton on the Canadian east coast: implications for aquaculture. Bull. Aquacult. Assoc. Canada 97-3: 9-18.

Bates, S.S. 1998. Ecophysiology and metabolism of ASP toxin production, p. 405-426. In: Anderson, D.M., A.D. Cembella, and G.M. Hallegraeff [eds.]. Physiological ecology of harmful algal blooms. Springer-Verlag, Heidelberg.

Bates, S.S. and J. Worms [eds]. 1989. Proceedings of the first Canadian workshop on harmful marine algae, Gulf Fisheries Centre, Moncton, New Brunswick, September 27 - 28, 1989. Can. Tech. Rep. Fish. Aquat. Sci. 1712: 60 p.

Blaylock, R. L. 1994. Excitotoxins - the taste that kills. Health Press, Santa Fe, New Mexico. 300 p.

Cembella, A.D. and E.C.D. Todd. 1993. Seafood toxins of algal origin and their control in Canada, p. 129-144. In: I.R. Falconer [ed.] Algal toxins in seafood and drinking water. Academic Press, London.

Couturier, C. 1988. Shellfish toxins aplenty. Bull. Aquaculture Assn. Can. 88-2: 11-27.

Forbes, J.R. [ed.]. 1991. Pacific coast research on toxic marine algae. Can. Tech. Rep. Hydrogr. Ocean Sci. 135: 76 p.

Forbes, J.R. [ed.]. 1994. Proceedings of the fourth Canadian workshop on harmful marine algae, Institute of Ocean Sciences, Sidney, British Columbia, May 3 - 5, 1994. Can. Tech. Rep. Fish. Aquat. Sci. 2016: 86 p.

Fryxell, G.A. and M.C. Villac. 1999. Toxic and harmful marine diatoms, p. 419-428. In: E.F. Stoermer and J.P. Smol [eds.] The diatoms. Applications for the environmental and earth sciences. Cambridge Univ. Press, Cambridge.

Gordon, D.C., Jr. [ed.]. 1991. Proceedings of the second Canadian workshop on harmful marine algae, Bedford Institute of Oceanography, Dartmouth, Nova Scotia, October 2 - 4, 1990. Can. Tech. Rep. Fish. Aquat. Sci. 1799: 66 p.

Gulland, F. et al. 1999. Unusual marine mammal mortality event – Domoic acid toxicity in California sea lions (*Zalophus californianus*) stranded along the central California coast, May – October 1998. NOAA Technical Memo NMFS-ORP-8, National Marine Fisheries Service, US Dept. Of Commerce, Silver Spring, Maryland.

Gunner, S.W. 1990. Mussel poisoning in Canada. J. Toxicol., Toxin Rev. 9: 89.

Hall, S. 1988. PSP and other shellfish toxins: recent outbreaks and research. J. Shellfish Res. 7: 565.

Hall, S. 1991. Natural toxins. In: D.R. Ward and C. Hackney [eds.] Microbiology of marine food products, p. 301-330. Van Nostrand Reinhold, New York.

Hallegraeff, G.M. 1993. A review of harmful algal blooms and their apparent global increase. Phycologia 32: 79-99.

Horner, R.A., D.L. Garrison, and F.G. Plumley. 1997. Harmful algal blooms and red tide problems on the U.S. west coast. Limnol. Oceanogr. 42: 1076-1088.

Hynie, I. and E.C.D. Todd [eds.]. 1990. Proceedings of a symposium on domoic acid toxicity. Can. Diseases Weekly Rep. 16 S1E: 123 p.

Jellett, J.F. 1993. Phytotoxins and shellfish aquaculture. World Aquacult. 24: 32-43.

Kizer, K.W. 1994. Domoic acid poisoning. Western J. Med. 161: 59-60.

MacKenzie, D. 1988. Mystery of mussel poisoning deepens in Canada - as the chain of death spreads to whales. New Scientist 117: 30.

Olney, J.W. 1994. Excitotoxins in foods. Neurotoxicol. 15: 535-544.

Pan, Y., S.S. Bates, and A.D. Cembella. 1998. Environmental stress and domoic acid production by *Pseudo-nitzschia*: a physiological perspective. Natural Toxins 6: 127-135.

Penney, R.W. [ed.]. 1996. Proceedings of the Fifth Canadian Workshop on Harmful Marine Algae. Can. Tech. Rep. Fish. Aquat. Sci. 2138: 195 p.

Quilliam, M.A. 1995. Seafood toxins. J. AOAC Internat. 78: 144-148.

Quilliam, M.A. and J.L.C. Wright. 1989. The amnesic shellfish poisoning mystery. Anal. Chem. 61 (18): 1053A-1060A.

Ravn, H. 1995. Amnesic Shellfish Poisoning (ASP). HAB Publications Series, IOC Manuals and Guides No. 31, Vol. 1, UNESCO, Paris, 15 p.

Richardson, K. 1997. Harmful or exceptional phytoplankton blooms in the marine ecosystem, p. 301-385. In: Blaster, J.H.S. and A.J. Southward [eds.] Advances in Marine Biology, Vol. 31. Academic Press Ltd., New York.

Scholin, C.A., F. Gulland, G.J. Doucette, S. Benson, M. Busman, F.P. Chavez, J. Cordaro, R. DeLong, A. De Vogelaere, J. Harvey, M. Haulena, K. Lefebvre, T. Lipscomb, S. Loscutoff, L.J. Lowenstine, R.I. Martin, P.E. Miller, W.A. McLellan, P.D.R. Moeller, C.L. Powell, T. Rowles, P. Silvagnl, M. Silver, T. Spraker, and F.M. Van Dolah. 2000. Mortality of sea lions along the central coast linked to a toxic diatom bloom. Nature 403: 80-84.

Sierra Beltrán, A., M. Palafox-Uribe, J. Grajales-Montiel, A. Cruz-Villacorta, and J.L. Ochoa. 1997. Sea bird mortality at Cabo San Lucas, Mexico: evidence that toxic diatom blooms are spreading. Toxicon 35: 447-453.

Smayda, T.J. 1990. Novel and nuisance phytoplankton blooms in the sea: evidence for a global epidemic, p. 29-40. In: E. Granéli, B. Sundström, L. Edler, D.M. Anderson [eds.] Toxic marine phytoplankton. Elsevier Sci. Publ. Co., Inc., New York.

Therriault, J.-C. and M. Levasseur [eds.]. 1992. Proceedings of the third Canadian workshop on harmful marine algae, Maurice Lamontagne Institute, Mont-Joli, Quebec, May 12 - 14, 1992. Can. Tech. Rep. Fish. Aquat. Sci. 1893: 154 p.

Todd, E. 1990. Amnesic shellfish poisoning - a new seafood toxin syndrome, p. 504-508. In: E. Granéli, B. Sundström, L. Edler, D.M. Anderson [eds.] Toxic marine phytoplankton. Elsevier Sci. Publ. Co., Inc., New York.

Todd, E.C.D. 1993. Amnesic shellfish poisoning - a review. J. Food Protection 56: 69-83.

Waldichuk, M. 1989. Amnesic shellfish poison. Mar. Pollut. Bull. 20: 359-360.

Whyte, J.N.C., J.C. Davis, and J.R. Forbes, Jr. 1997. Harmful algae in Canadian waters and management strategies. Bull. Aquacult. Assoc. Canada 97-2: 161-171.

Wood, A.M. and L. Shapiro [eds.]. 1993. Domoic acid: final report of the workshop, Oregon Institute of Marine Biology, February 21-23, 1992. Oregon Sea Grant, ORESU-W-92-003.

Wood, A.M., L. Shapiro, and S.S. Bates [eds.]. 1994. Domoic acid: final report of the workshop, Oregon Institute of Marine Biology, February 21-23, 1992. Oregon Sea Grant, ORESU-W-94-001 (second edition).

Wright, J.L.C. 1989. Domoic acid, a new shellfish toxin: the Canadian experience. J. Shellfish Res. 8: 444.

Wright, J.L.C. 1998. Domoic acid - ten years after (guest editorial). Natural Toxins 6: 91-92.

1.2.16 REFERENCES

U. S. Army Corps of Engineers. 1987. *Reservoir Water Quality Analysis*, Engineering Manual, EM-1110-2-1201. U.S. Army Corps of Engineers, Washington, DC.

U. S. Environmental Protection Agency. 1993. *Guidance Specifying Management Measures for Sources of Nonpoint Pollution in Coastal Waters*, 840-B-92-003, U.S. Environmental Protection Agency, Washington, DC.

Walker, W. W., Jr. 1981. “Empirical methods for predicting eutrophication in impoundments. Phase I: Database development,” Technical Report E-81-9, prepared by William W. Walker, Jr., Environmental Engineer, Concord, MA, for the US Army Engineer Waterways Experiment Station, Vicksburg, MS.

1.3 OVERVIEW OF LIMNOLOGICAL PROCESSES IN TAILWATERS

1.3.1 INTRODUCTION

Reservoir tailwaters may be considered a unique type of lotic (riverine) aquatic system since the quantity and quality of the source is a function of an upstream control structure (e.g., dam). The impoundment of the river results in water quality changes and regulation of flow, both of which impact water quality processes in the downstream area or tailwater. Seasonal and operational influences are observed in the tailwater and are often reflected as spatial gradients as well. Thus a tailwater may be defined as that region downstream from a dam that reflects water quality and flow conditions directly influenced by the impoundment and regulation of the river. Often, the "tailwater" is limited to a reach defined by a particular study or features such as downstream dams, secondary tributaries, or point sources that greatly influence the water quality or hydrology.

The regulation of flow in a reservoir tailwater is a function of the operation of the dam which was built for purposes such as flood control, navigation, hydropower production, water supply, and recreation. Operation for specific project purposes results in different release regimes or hydrographs. For example, the high flow of a runoff event may be "attenuated" by the operation of the dam resulting in a decrease in the peak discharge occurring for a longer duration, "absorbed", resulting in a constant outflow, or manipulated as a staged discharge (Figure 1.3.1). Each of these hydrographs establishes wetted areas, depths, currents, resuspension and transport processes, and mixing zones that define the physical conditions for biological and chemical processes. The extent of impact is a result of discharge level and duration. During the high flow discharge, wetted areas, depths, and currents are at a maximum throughout the tailwater area. Maximum chemical concentrations occur on the leading edge of the hydrograph (as observed for inflow hydrographs), and coincident with resuspension and transport conditions (Ashby et al., 1995; Barillier, et al., 1993). Typically, water quality in the tailwater reflects near steady-state conditions, spatial and/or temporal changes (dependent upon the constituent), or conditions in the upstream reservoir (Nix et al., 1991). Channel morphometry, flow dynamics, secondary inflows, and point/nonpoint sources are also major contributors to the water quality of the tailwater. The length of the tailwater required for any constituent to return to values close to those measured at the inflow to an impoundment, or to achieve a new equilibrium may be considered as the "recovery distance" (Palmer and O'Keefe 1990). Typically, recovery distance is directly proportional to river size.

General features of reservoir tailwaters are summarized in Figure 1.3.2. In this figure, the velocity of the release from the dam is high and processes such as mixing, aeration, sediment transport, and degassing occur downstream at a riffle zone. Velocity and currents are varied further downstream in the vicinity of an island and additional mixing occurs at the confluence of a secondary tributary. Sediment resuspension and transport also occur at areas with high

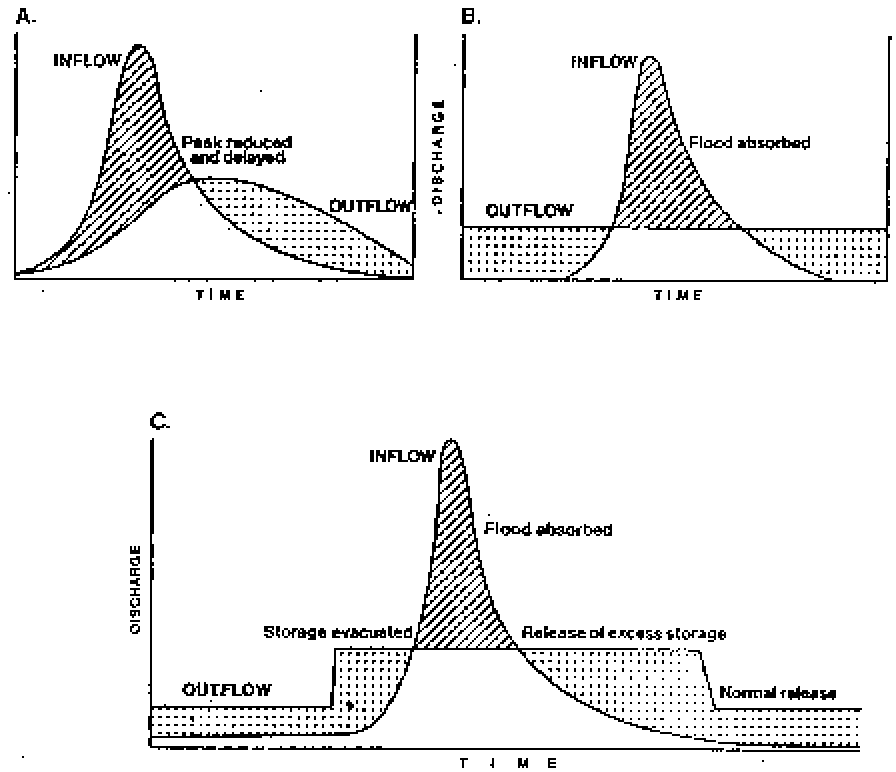


Figure 1.3.1 Three major types of flow regulation at reservoirs: A) attenuation, B) storage, and C) manipulation. (From *Impounded Rivers,* G. E. Petts, 1984, Copyright John Wiley & Sons Limited. Reproduced with permission.)

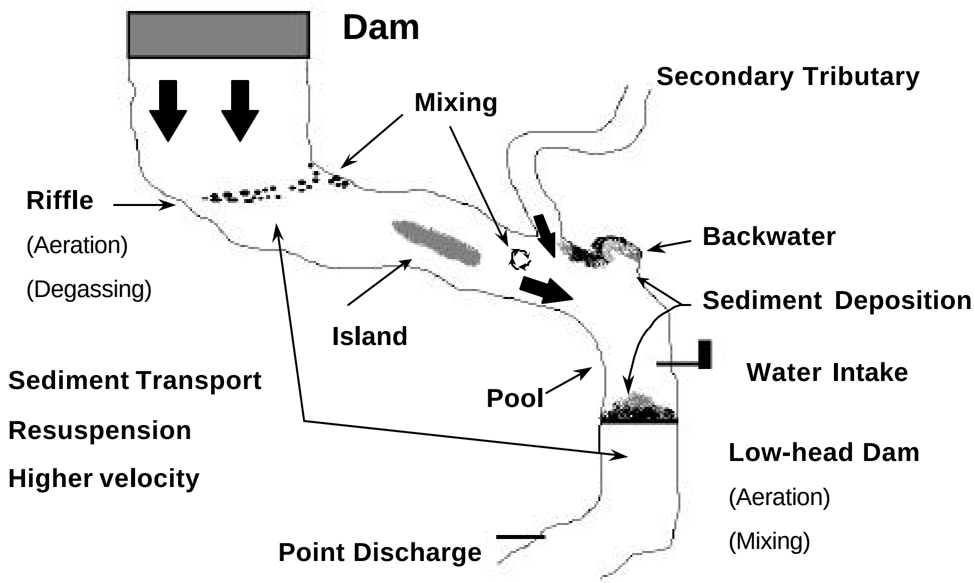


Figure 1.3.2 Idealized reservoir tailwater depicting morphometric features (e.g. riffles and pools), downstream users, material transport/deposition scenarios, and flow related influences.

velocities. The development of a backwater area and sediment deposition is depicted downstream from the confluence of the secondary tributary. Further downstream a low-head dam has been constructed forming a pool for a water intake and resulting in decreased velocity and increased sediment deposition. The low-head dam also provides aeration, mixing, increased velocity, and increased sediment transport as water passes over the dam. A point discharge downstream from the low-head dam is also shown indicating that specific sources of constituents also exist in tailwaters.

Physical changes due to impoundment, hydrology, and morphology are primarily reflected by a change in velocity, temperature, amount and distribution of suspended solids (and hence, available light), and concentrations of dissolved gases. Chemical changes are most often the result of water quality conditions in the upstream impoundment, biological processes in the tailwater, and physicochemical changes in the water column (including air/water and water/channel bottom interfaces). Examples of chemical changes include the increase in concentrations of nutrients and reduced elements from anoxic hypolimnia, decreases in dissolved oxygen concentrations with distance due to biological oxygen demand of organic matter from the upstream impoundment, precipitation reactions associated with oxidation of reduced metals, degassing, and sorption/desorption reactions on substrates.

While often considered to adversely impact the downstream habitat, positive effects of dam construction can also occur and will be included in the following discussion. In a broader context, Palmer and O'Keefe (1990) suggest that reservoirs receiving agricultural and urban runoff can improve conditions in the downstream area, emphasizing the relationships to the catchment area. The objective of this section is to provide an overview of water quality processes in reservoir tailwaters and their relationships to project operations. While this discussion focuses on processes downstream from selected dams, the logical expansion to the basin or river level with multiple impoundments has been initiated by Ward and Stanford (1983) and referred to as the Serial Discontinuity Concept which is modeled after the River Continuum Concept of Vannote et al. (1980). More detailed information is available in the literature and excellent references include Hynes (1970) and Petts (1984).

1.3.2 HYDROLOGY/MATERIAL TRANSPORT

The movement of water or hydrology in a reservoir tailwater greatly affects the transport of material. Flow patterns vary from laminar (parallel flow at a constant low speed) to turbulent (characterized by irregularity of flow moving in different directions and speeds) flow as velocity increases (Figure 1.3.3). Turbulent flow is an erratic and mixing progression of water and describes most of a stream's flow pattern. As the slope of the river channel increases, accelerating forces are checked by turbulence and channel roughness which induces turbulence, and velocity is limited. The boundary between the smoothness of laminar flow and the eddying

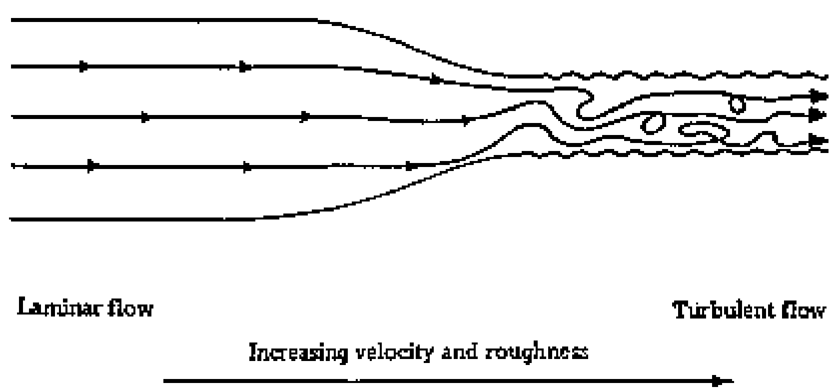


Figure 1.3.3 General diagram of laminar and turbulent flow

sinuosity of turbulent flow is referred to as the Reynold's number (N_R) which is dimensionless. In the stream channel it is the value derived from dividing the product of the mean grain diameter (ñ), the specific discharge (q), and the fluid density (d) by the dynamic viscosity (µ) and is effected by temperature affects on density and viscosity.

The equation for the Reynold's number may be represented as:

$$N_R \; ' \; \tilde{n} \frac{qd}{\mu}$$

Reynold's numbers greater than 2800 indicate turbulence or low viscosity and are much more prevalent than numbers for laminar flow, i.e less than 2000.

Laminar flow is significant in streams to plants and animals (e.g., refuge) and is found as a thin layer on river substrate such as gravel/rocks and woody debris. Turbulent flow is effective at eroding the stream channel and transport of materials. Zones of maximum turbulence are associated with changes between forward flow (river bends) and the friction of the stream channel (riffle zones). The Austausch (German meaning exchange) coefficient (A) is a measure of turbulence or describes the mixing in addition to the mixing associated only with molecular diffusion. It is an attempt to quantify all the aimless movements that are counter to the main direction of flow (Cole 1983). Turbulence and laminar flow impact the concentration of dissolved gases and dissipation of heat (both increase with an increase in turbulence).

Flow pattens downstream from a hydropower project are presented in Figure 1.3.4 which is a photograph of the release from a scaled, physical model with markers used to depict flow patterns. As represented by the light markers, there are distinct flow patterns and areas of different velocities that become less defined as flow proceeds downstream. High velocity, mixing, and turbulence are indicated in the lower right corner, just downstream from the powerhouse. There is a well-established, counter-clockwise flow established further downstream that creates increased turbulence and a smaller clockwise flow in the vicinity of the flood control gates (lower left corner). These patterns are representative for reservoir tailwaters but would vary as a function of release regimes and channel morphometry.

Velocity (distance/unit time) is determined by the quantity of water, channel shape, bottom texture/bank structure, and gradient or slope (Table 1.3.1). Wide, deep rivers with a slight slope would have a greater velocity than a small, shallow stream on a much greater slope. Flow also proceeds at a faster rate on smooth channels than it does on rough-bottom channels and increases at constrictions. Velocity varies in a stream channel with greater velocities occurring on the surface in the center of the stream (Figure 1.3.5).

Discharge is the quantity of water that passes a particular point in a unit of time and is represented by the following equation:

Discharge (m^3 s^{-1}) = channel width (m) * channel depth (m) * water velocity (m s^{-1})

Discharge varies across the cross section of a channel as a function of velocity and morphometry (Figure 1.3.5), therefore, measurement of discharge is often conducted with the use of subsections (Figure 1.3.6). Measurements of velocity are conducted at a depth located 0.6 units

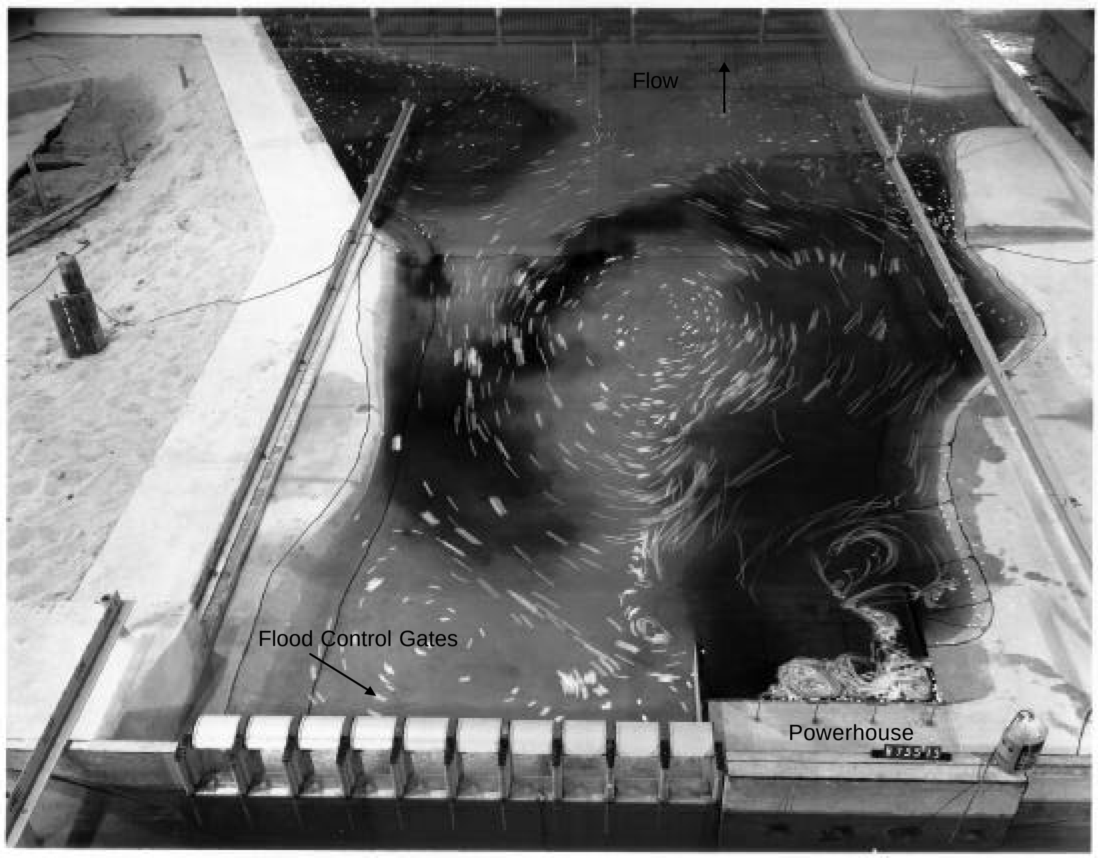


Figure 1.3.4 Physical model simulation of currents downstream from a hydropower dam

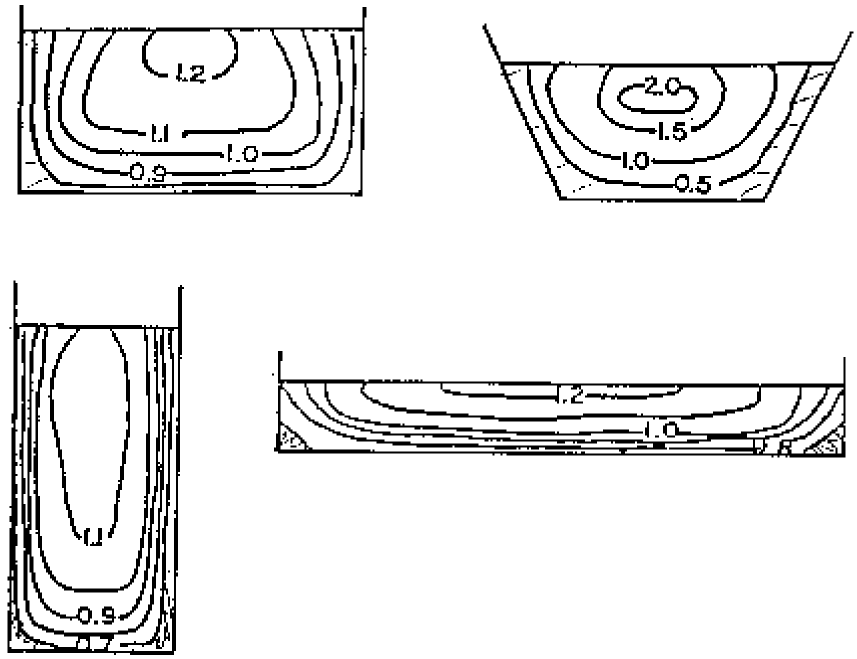


Figure 1.3.5 Idealized velocity distributions in channels of different slopes

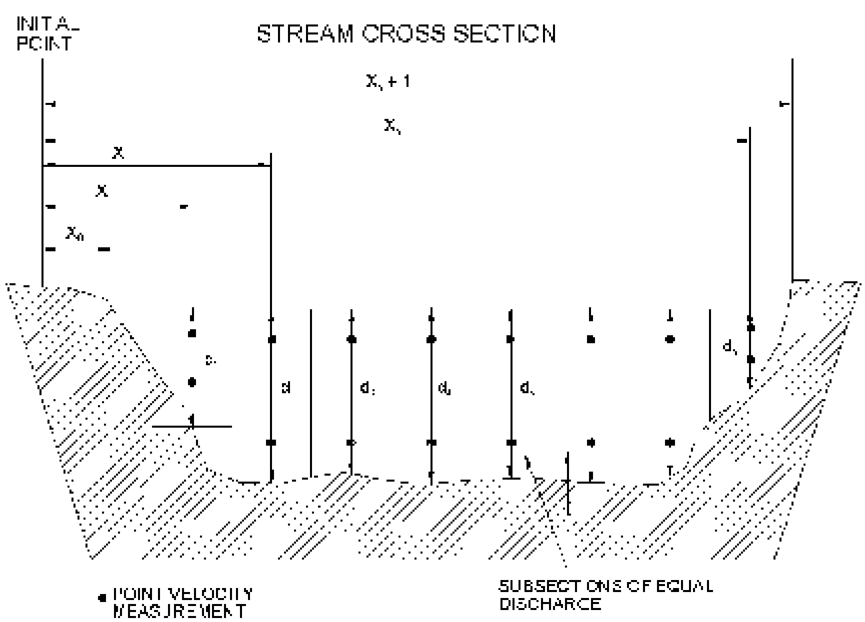


Figure 1.3.6 Subsections of stream cross section for velocity measurements

from the surface in shallow streams and at 0.2 units from the surface and bottom in deeper streams. Total discharge is then summarized over the entire cross section by adding partial discharges calculated for each subsection.

Material transport refers to materials transported as either dissolved substances, suspended solids, or as the bed load. Dissolved substances will be discussed later. Suspended solids transported in a stream are a function of the weight of the suspended material (often represented by grain size due to analytical techniques) and velocity (Table 1.3.2). Materials in movement along a stream bottom constitute the stream's bed load. The amount of suspended material in a stream impacts physical characteristics of the stream water such as heat adsorption and penetration/scattering of light. Measurement of material transport is often conducted by direct measurement of total, suspended, and dissolved solids and indirectly using measurements of turbidity (light scattering by particulates).

Material transport is related to the type of substrate which is a function of the area geology and hydrologic conditions. Substrates vary from eroded bedrock to fine clays and channels can exhibit combinations of materials sized between boulders and clays. The hydraulics, morphology, and substrate availability usually result in a sorting of substrate by size throughout the channel. For instance, heavier material will settle in areas of decreased flow such as deep areas, banks, and bends. The permanency of this sorting is subject to the magnitude of flow (Table 1.3.2).

The hydrology and material transport associated with reservoir operations greatly influences the water quality in reservoir tailwaters. For instance, a flood control structure will retain high flow while maintaining a controlled discharge. This type of operation greatly influences retention time (Figure 1.3.7) and provides for particulate settling in the reservoir, decreasing transport to the tailwater, and relatively “steady-state” conditions in the tailwater during the high flow release. This is most often true when high flow events have appreciably impacted reservoir water quality resulting in near-homogeneous conditions. After flood control releases have been curtailed, routine releases are often reflective of the inflow hydrograph and a more “natural” hydrograph is established with biological and chemical conditions reflective of reservoir release water quality and riverine processes. During low flow periods, only a minimum flow, usually very similar to the inflow, is maintained in the release. Minimal flow often occurs when project operations, such as recreation in the upstream reservoir, require a stable or established surface elevation or during periods of low flow.

Hydropower production results in a more structured hydrograph (Figure 1.3.8a) than flood control hydrographs (Figures 1.3.8b and 1.3.8c). A hydropower project also routinely adds a more detailed temporal component based on demand for electricity (Figure 1.3.9) and often impacts daily (diurnal) temperature and dissolved oxygen cycles (Jabour et al. 1996; Ashby et al. 1995). Daily heating and photosynthesis are often disrupted as a result of increased discharge during “peak” demands for electricity and new conditions are established that reflect discharge water quality (Figure 1.3.10). For example, pre-generation temperatures and dissolved oxygen

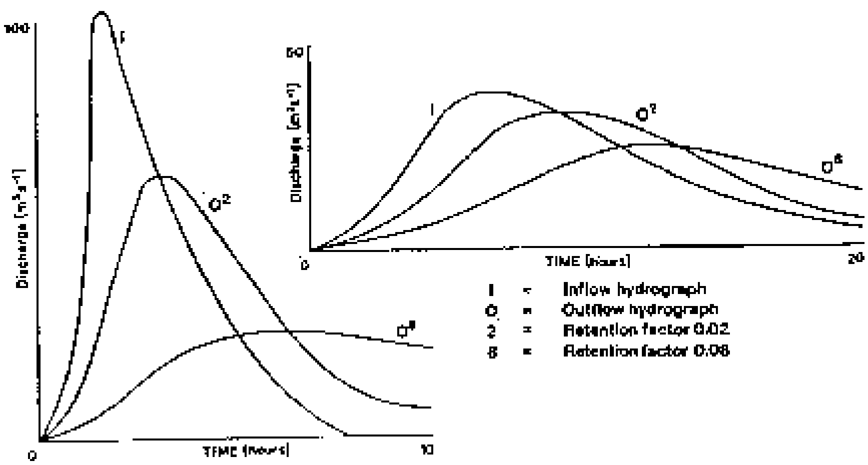


Figure 1.3.7 Tailwater hydrographs in response to different retention times for a rapid and high flow event and an event of lesser magnitude but longer duration. (From *Impounded Rivers*, G. E. Petts, 1984, Copyright John Wiley & Sons Limited. Reproduced with permission)

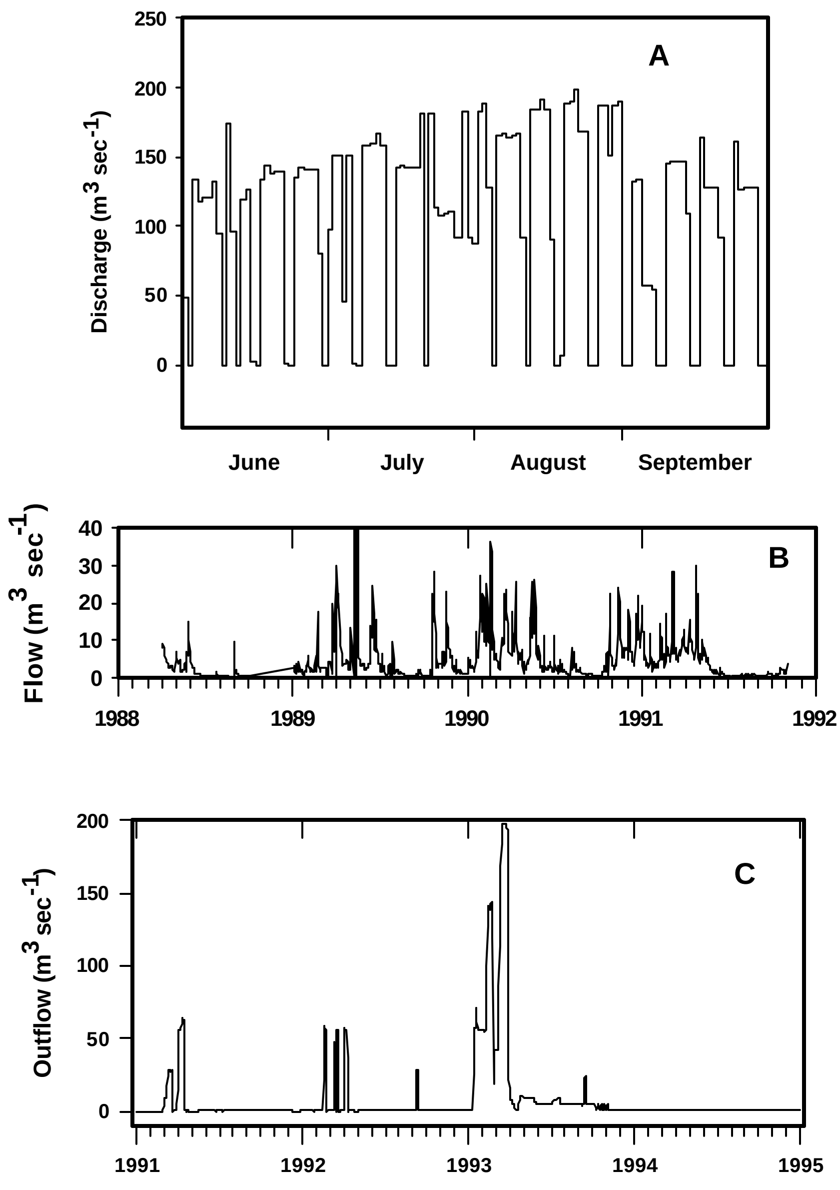


Figure 1.3.8 Hydrographs from (A) a hydropower operation (mean daily discharge), (B) a flood control project in New York, and (C) a flood control project in Arizona.

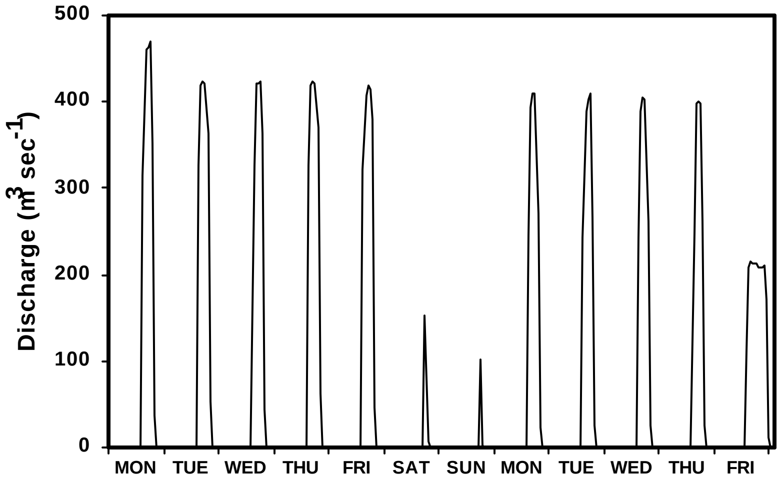


Figure 1.3.9 Hourly discharge from a peaking, hydropower operation.

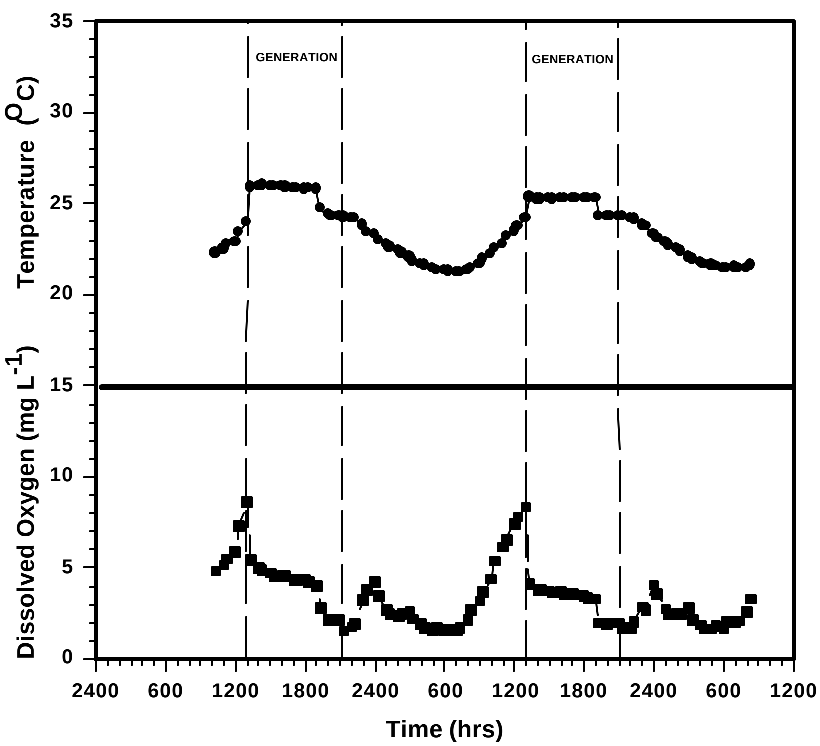


Figure 1.3.10 Temperature (upper panel) and dissolved oxygen concentrations (lower panel) before, during, and after peaking hydropower generation at the tailwater of Lake Texoma, Oklahoma and Texas. Vertical dashed lines depict the start and stop times for generation. The decrease in temperature and dissolved oxygen near the end of generation reflect changes in water quality in the withdrawal zone when one of the two turbines were shut down

concentrations are increasing in the morning until generation establishes a relatively constant temperature near 26 ^{O}C and decreased dissolved oxygen concentrations near 5 mg L^{-1}. Once generation stops (around 9:00 to 10:00 p.m. after the peak in demand) temperature and dissolved oxygen concentrations return to pre-generation patterns until the onset of the next generation cycle. Of interest in Figure 1.3.10, is the increase in dissolved oxygen concentration around 11:00 p.m. which may be a function of reaeration associated with exposed riffle zones as the river stage drops coincident with the decrease in discharge. Temperatures and dissolved oxygen concentrations may be quite different depending on the conditions in the withdrawal zone in the upstream impoundment, especially during summer stratification. Alternatively, winter releases may result in warmer temperatures with adequate dissolved oxygen concentrations in reservoirs located in northern latitudes.

Chemical changes are often reflected as decreased concentrations near the dam (attributed to dilution of hypolimnetic water with mid-depth or surface water) during generation with gradual increases or decreases in total concentrations occurring in the tailwater as a result of resuspension or settling, respectively, associated with discharge rates. These changes will be discussed in more detail later in this section.

Regardless of project purposes, the discharge in a reservoir tailwater is a function of water availability and quite often only a minimum flow is maintained. Recently, augmentation of minimum flows has been viewed as an opportunity to improve tailwater habitats and water quality. Augmentation of minimum flow is usually accomplished with changing reservoir operating guidelines for storage and release. Quite often this is accomplished via a more frequent review of water allocation and results in minimal impacts to project purposes. The net result of flow augmentation is a more stable aquatic habitat, improved water quality via dilution of downstream inputs, increased navigation capabilities, and increased supply for downstream users.

1.3.3 MORPHOLOGY

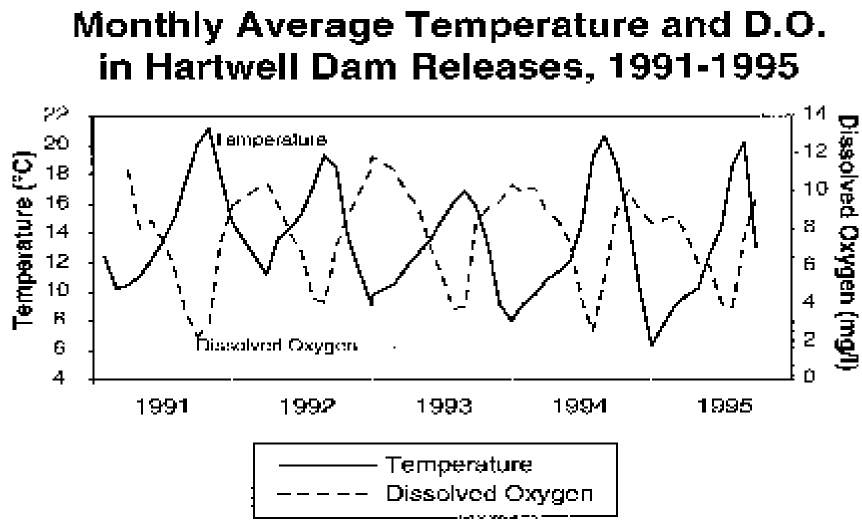
Rivers and tailwaters follow various courses as they progress seaward (from high altitude to low altitude) as determined by the geologic setting. Consequently, they display a variety of shapes and channel configurations that effect flow patterns and material transport as described above. The channel morphometry downstream of a reservoir may have been modified, particularly for navigation and flow, and is almost always different from conditions existing before the construction of the dam. Major morphometric features important to water quality processes in tailwaters include such things as bank erosion potential, riffles and pools, backwater areas, stream bed composition, and hydraulic features that would affect currents and water movement. Bank erosion would increase sediment transport and alter habitats. Riffles and pools, backwater areas, stream bed composition and hydraulic features would affect habitats of aquatic organisms and may contribute to spatial gradients in water quality. Sear (1995) suggests that changes in the sediment transport regime in the tailwater of a hydropower project are controlled by the hydraulics of the channel morphometry (riffles, pools, etc.) and contribute to the degradation of habitat for fish spawning and changes in riparian areas. Due to changes in energy associated with velocity, riffles would tend to act as areas of transport and erosion of particulate matter whereas pools would serve as areas of deposition unless velocities were sufficient (e.g., high discharge) to provide resuspension of particulates. Water quality, which is related to particulate matter, would also be affected by changes in sediment transport regimes.

1.3.4 THERMAL PATTERNS

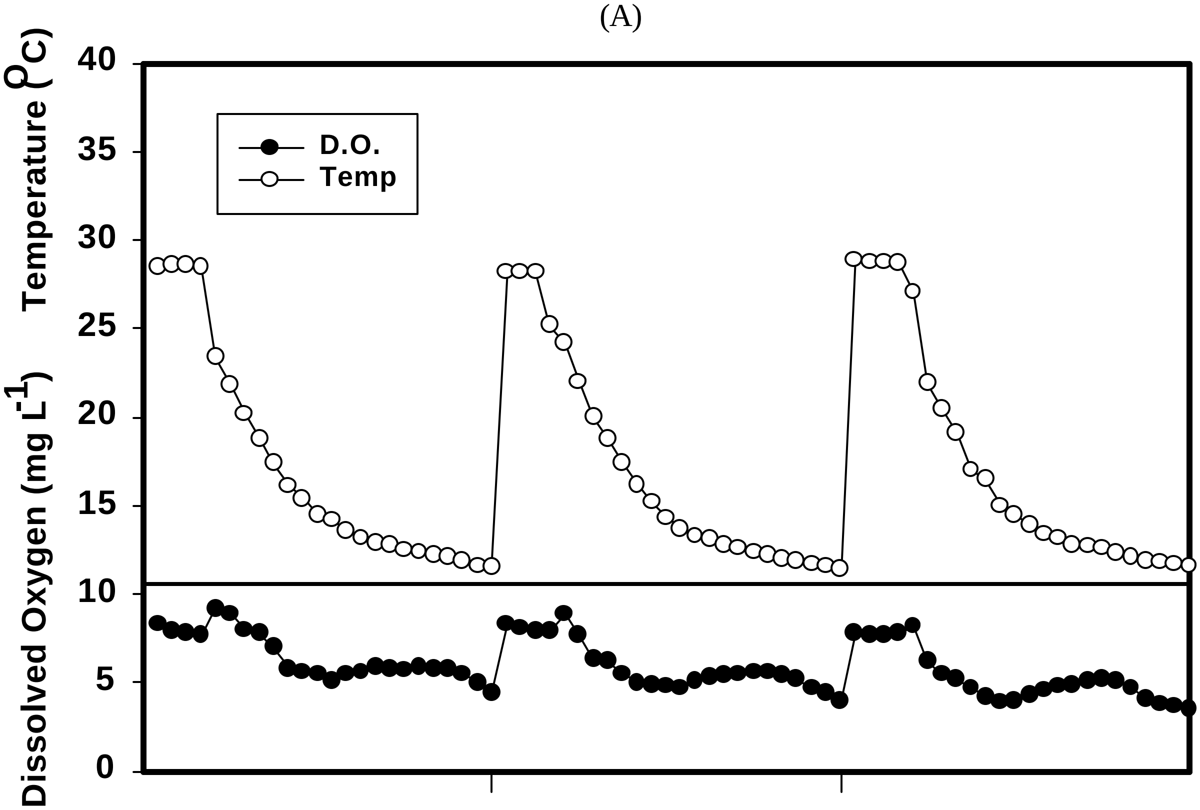
Temperatures in tailwaters are a function of daily and seasonal cycles in solar heating and release operations at the dam (Figure 1.3.11). In the northern hemisphere, maximum temperatures occur in July through September, minimum temperatures occur in January and February, and are often very similar over a period of years (Figure 1.3.11a). For hydropower operations, daily ranges in temperature can be quite extreme (e.g. 27 to 12 ^{O}C, Figure 1.3.11b, upper panel). Variability is often reflective of latitude, altitude, and project depth and operation as well as extremes in annual climate. Release temperatures can be manipulated by changing intake levels in the outlet works or employing alternative release strategies such as bottom releases or spills (release through flood control gates) (Figure 1.3.12). Annual or seasonal temperature management strategies may be developed, primarily for fisheries requirements, using selective withdrawal for maintaining downstream temperatures during critical periods such as spring spawning and fall migration seasons. However, changes in operations to develop desired conditions in the tailwater may result in an adverse impact, such as a decrease in available habitat, in the upstream impoundment (Hudgins 1995).

1.3.5 DISSOLVED GASES

The primary dissolved gases of concern are dissolved oxygen, nitrogen, and carbon dioxide. In well aerated releases and releases from aerobic areas of the reservoir, favorable concentrations are maintained. Dissolved oxygen dynamics are most pronounced in releases from hypolimnia with low dissolved oxygen concentrations and projects with varied operations (i.e. hydropower as depicted in Figure 1.3.13). In this figure and in Figures 1.3.10 and 1.3.11b, minimum dissolved oxygen concentrations occur coincident with hydropower generation. Seasonally, lower values occur during maximum thermal stratification in the upstream impoundment (Figure 1.3.11a). As these release waters move downstream, turbulence, reaeration, and primary productivity increase the dissolved oxygen concentration (Figure 1.3.14). However, oxygen demand may be increased due to metabolism of carbon (from increased hypolimnetic concentrations). Generally, a recovery to near equilibrium concentrations occurs at some point downstream. Critical periods for dissolved oxygen occur primarily during late summer stratification and the establishment of anoxic conditions in the withdrawal zone when concentrations in the releases are often low (see Figure 1.3.11a). There is often a diurnal cycle to oxygen production and consumption in productive systems (Figure 1.3.15) that may be disrupted by reservoir operations (see Figures 1.3.10 and 1.3.13). For example, in Figure 1.3.10, dissolved oxygen concentrations begin to increase at 6 a.m. and prior to hydropower generation and coincident with increasing light availability and continue to increase until hydropower generation near 1 p.m. completely flushes the system. Most dams provide some form of reaeration during release due to



(A)



(B)

Figure 1.3.11 Temperature and dissolved oxygen concentrations from the releases from Hartwell Dam (a peaking hydropower project on the Georgia and South Carolina border) depicting A) seasonal and annual cycles and B) daily cycles.

turbulence associated with tailrace structures (natural and man-made, channel roughness and outlet works (including modifications) and modified operations

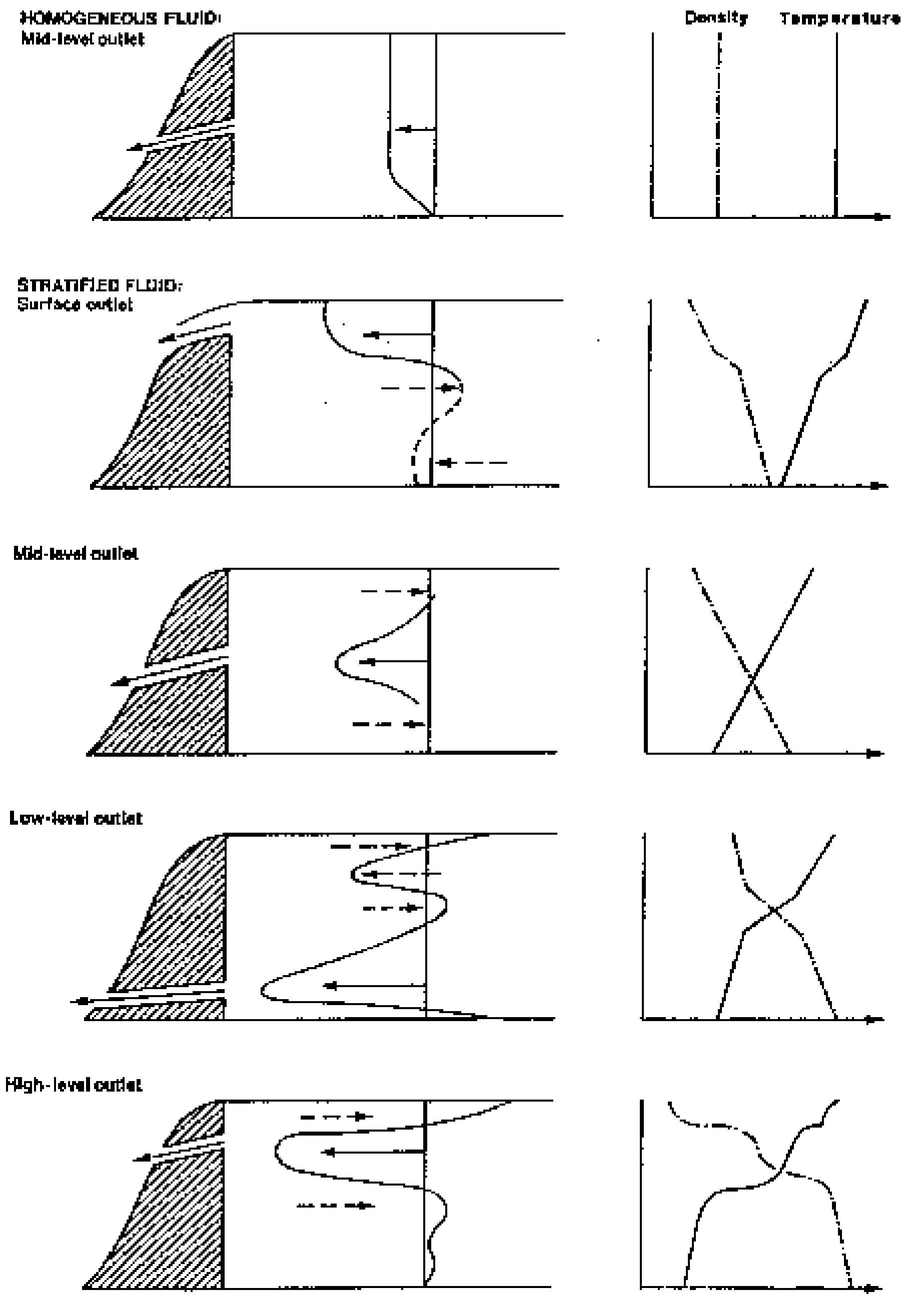
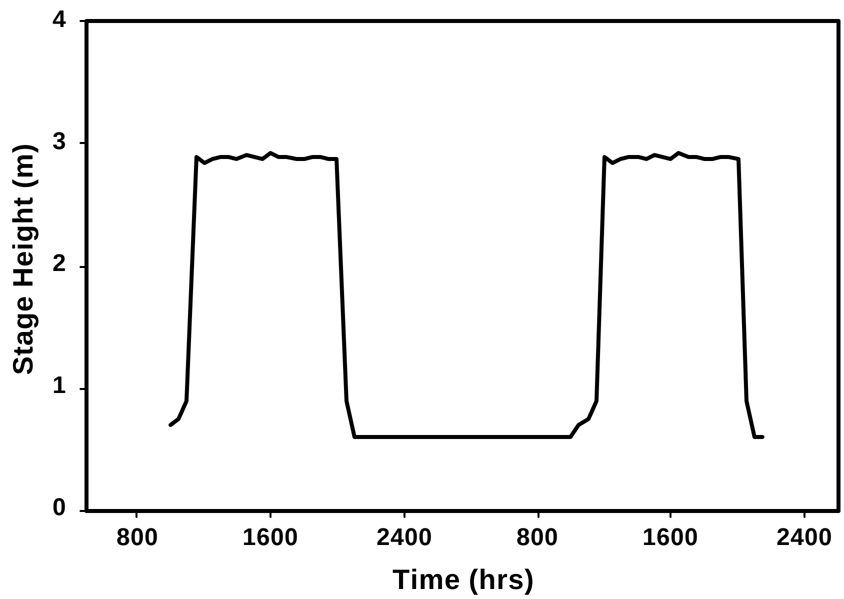
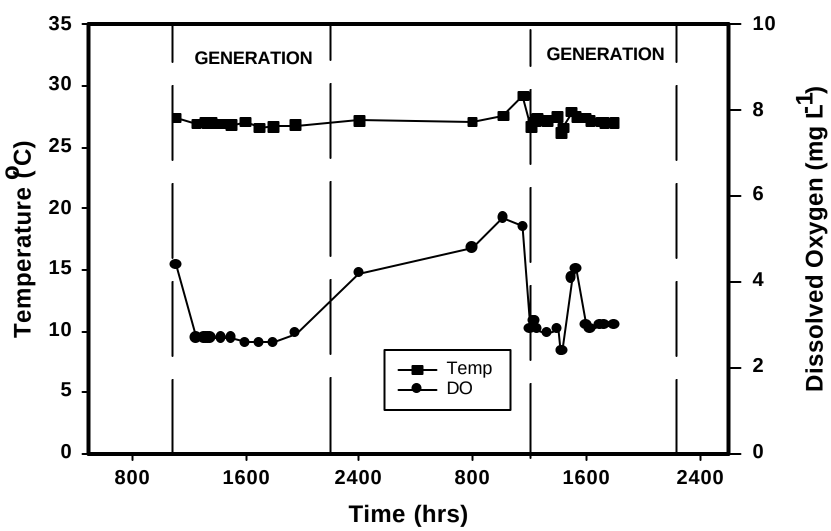


Figure 1.3.12 Effects of selective withdrawal from various depths upon flow patterns, density, and temperature profiles, in homogeneous and stratified lakes. (From *Impounded Rivers*, G. E. Petts, 1984, Copyright John Wiley & Sons Limited. Reproduced with permission)



(a)



(b)

Figure 1.3.13 Temperature and dissolved oxygen concentrations in response to peaking hydropower generation in (figure b provided for reference) the tailwater of West Point Lake, Georgia and Alabama. Note the diurnal warming and increase in dissolved oxygen which is disrupted by the onset of generation as represented by change in stage.

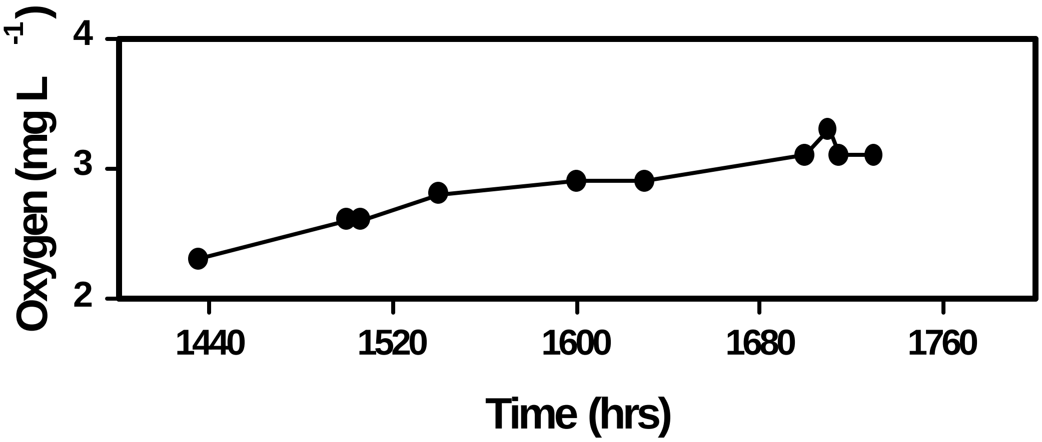


Figure 1.3.14 Change in dissolved oxygen concentrations in the release waters from West Point Dam as a function of travel time (distance)

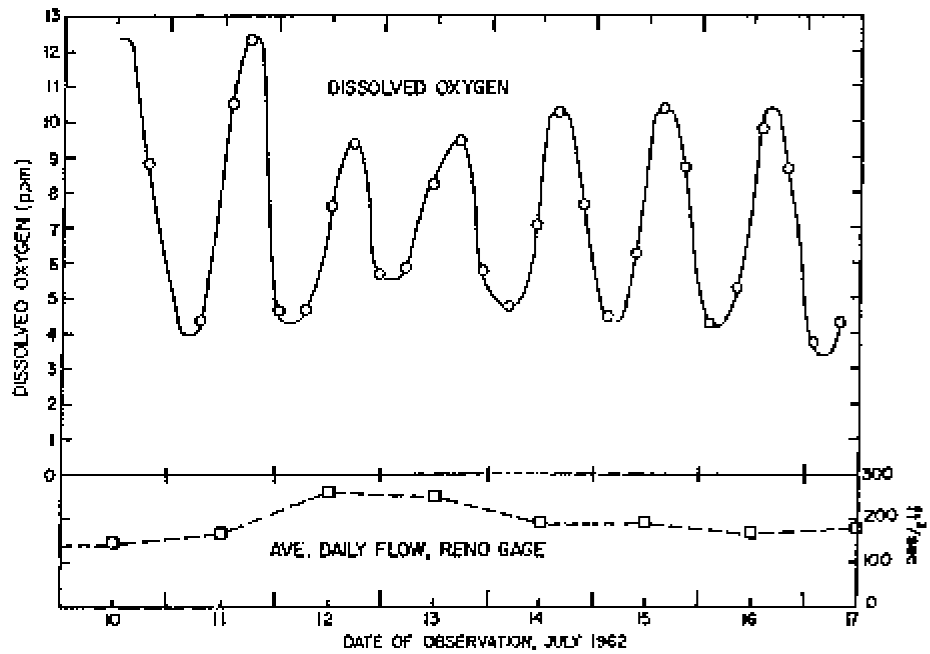


Figure 1.3.15 Dissolved oxygen and flow in the Truckee River (O'Connell et al. 1962). Note the diurnal cycle. (From Krenkel and Novotny (1980), reprinted with permission of Academic Press

such as minimal spills can be used to improve conditions during critical periods (Daniil et al. 1991).

Dissolved nitrogen in reservoir tailwaters is problematic when concentrations exceed saturation (i.e., supersaturation) and result in gas bubble disease in fish. Gas supersaturation occurs when air is entrained a water jet that plunges into a significant depth of water. The entrained air is transported as bubbles to the bottom of the stilling basin and the bubbles become dissolved in the water under hydraulic pressure. The magnitude of gas supersaturation depends upon the type of hydrostatic structures, magnitude of discharge, and depth of water in the stilling basin. Effects of nitrogen saturation on fish have been documented in studies conducted in the Columbia and Snake River Basins (Ebel et al. 1975; Dawley 1986). The severity of symptoms exhibited by fish exposed to high concentrations of dissolved nitrogen depends on the saturation level (usually above 110 percent), exposure time, water temperature, and the overall health of the fish. External symptoms include exopthalmia or pop-eye, bubbles in the caudal fin or hemorrhage, and emboli in gill blood vessels. Numerous studies conducted by the CE describing sampling techniques, equipment deployment, data collection and analyses, and predictive capabilities and be found in U.S. Army Corps of Engineers (1996).

Carbon dioxide is produced by respiration within tailwaters and concentrations may vary, especially between riffles and pools (Neel 1951). Typically, in pools, dissolved oxygen decreases while carbon dioxide concentrations increase as water moves through but equilibrium is restored as a function of turbulence associated with riffles. Increased carbon dioxide concentrations raises the carbonic acid content and lowers the pH (see discussion of the carbonate system in the overview of reservoir limnology). Carbon dioxide is either lost to the atmosphere or removed by the interactions with calcium carbonate. The equilibrium of carbon dioxide and calcium carbonate (especially in limestone-rich areas) provides a good buffer for the system and helps stabilize pH. In highly productive areas (areas with high photosynthetic rates), removal of carbon dioxide can result in deposition of calcium carbonate, especially in hardwater (i.e. high calcium concentrations) systems.

1.3.6 CHEMICAL PROCESSES

Chemical processes in reservoir tailwaters are a function of the quantity and quality of chemical constituents delivered from the upstream impoundment and reflect water quality conditions in the reservoir. Chemical processes of major concern usually involve nutrients (nitrogen and phosphorus), metals (iron and manganese), and hydrogen sulfide. Contaminants such as polychlorinated biphenyls (PCBs), mercury, pesticides are usually considered on a site specific basis. Nutrient concentrations are often elevated during summer stratification and with large runoff events. Monitoring is usually conducted to evaluate eutrophication processes such as algal, periphyton, and macrophyte production in the tailwater. Nutrient loading to downstream receiving water bodies is also a concern. Concentrations of certain species such as ammonia or nitrogen are also used in describing water quality for water supply. Processes involving metals are usually focused on oxidation of reduced iron and manganese. These metals contribute to taste, odor, and staining problems for downstream water supply intakes and provide precipitates and oxide coatings on streambed substrates that may impact the biota. Elevated concentrations of reduced metals may be toxic to the biota as well. Hydrogen sulfide is indicative of anoxia in the upstream reservoir and is primarily problematic as the degassing of dissolved sulfide as hydrogen sulfide. This gas is readily derived upon oxidation, quite noxious, and potentially toxic and corrosive at low concentrations.

Metals are often in elevated concentrations in release water especially from hypolimnetic releases. Iron, manganese, and sulfur (considered here with metals) are the constituents of most concern due to their increased mobility associated with oxygen depletion. Iron is readily oxidized upon release (exerting a chemical demand on dissolved oxygen). Oxidized iron forms a particulate (and may complex other metals and organics) and is transported or deposited in the downstream area as a function of physical processes described earlier. Recent investigations indicate that biological processes and substrate interactions may play an important role in removing iron from the release water. Manganese oxidation (more probably biological removal) is much slower and often function of substrate availability. Hydrogen sulfide is gaseous and is rapidly removed to the atmosphere (hence the rotten egg or sulfur smell in the immediate tailwater).

Examples of metals dynamics in reservoir releases are depicted in Figure 1.3.16 and Figure 1.3.17. Concentrations of total iron and manganese increase coincident with thermal stratification with maximum concentrations occurring in late summer and lower discharge. The temporal pattern for total manganese is less variable annually than the pattern for total iron. Figure 1.3.17 depicts dissolved and particulate (calculated as the difference between measured total and dissolved fractions) concentrations of iron and manganese. In general, iron is most often observed in the particulate form and manganese is most often observed as dissolved. This is typical for most reservoir releases and indicative of the higher oxidation reduction potential for iron. The higher oxidation reduction potential for iron allows for a higher oxidation rate and increased formation of particulates which may contribute to the variability in tailwater concentrations. The spatial dynamics of iron and manganese in reservoir tailwaters are a function of flow (Figure 1.3.18) and vary by site (Nix et al. 1991). In general, higher concentrations are observed further downstream during higher flow and are then observed to decrease with distance (triangles, Figure 1.3.18). Conversely, during lower flow, concentrations tend to decrease quickly for iron and almost linearly or as a first order process for manganese (circles, Figure 1.3.18). As observed for seasonal patterns, the response of total and dissolved iron compared to total and dissolved manganese is quite different with manganese remaining primarily in the dissolved fraction and iron occurring in the particulate (difference between total and dissolved) fraction.

Although reservoirs act as sediment and nutrient traps (sinks), mobilization of sedimentary nutrients during oxygen depletion can result in increased nutrient concentrations in release water.

Concentration increases are often seasonally distributed but can be a function of extreme high flow events when reservoir retention times are greatly reduced. Temporal distribution of nutrients in tailwaters may be described with observations of total nitrogen and total phosphorus. Total nitrogen concentrations remain relatively constant (Figure 1.3.19), however the contribution of ammonia and nitrate concentrations to the total would vary seasonally. During anoxia in the upstream impoundment, higher ammonia concentrations would

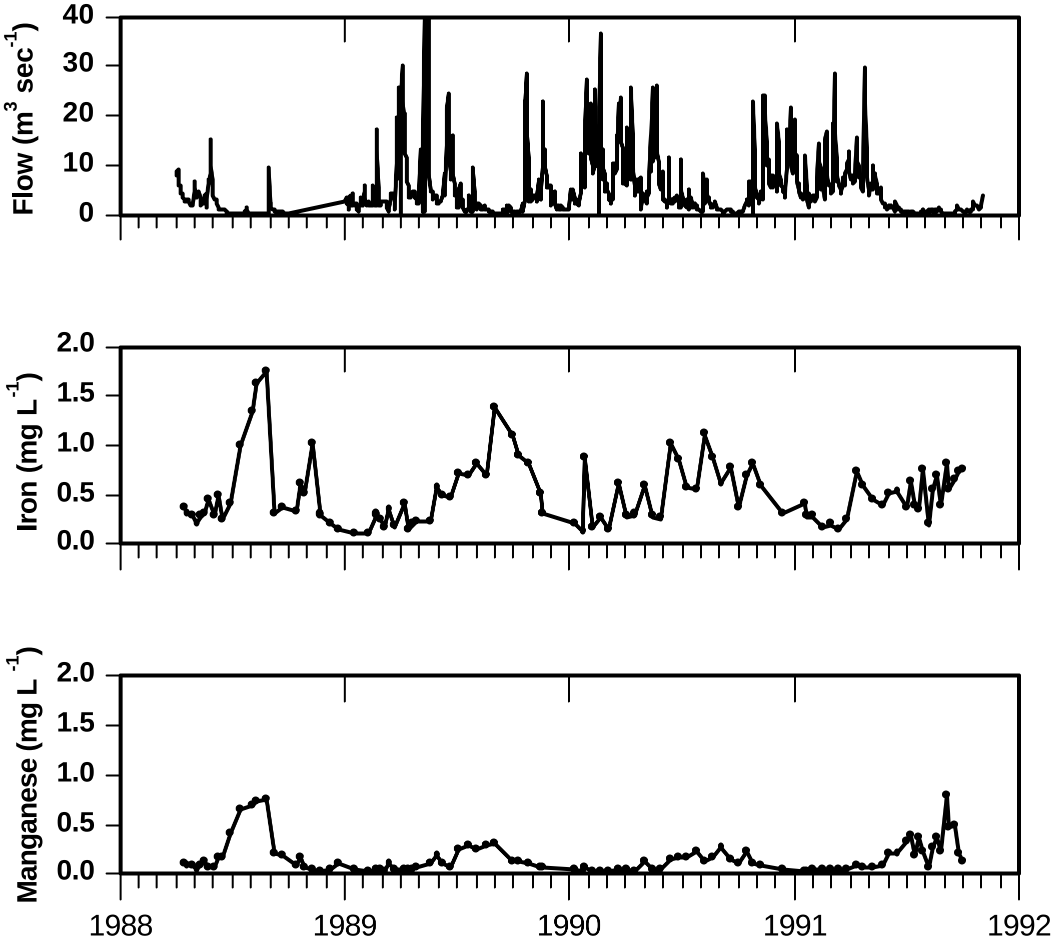


Figure 1.3.16 Temporal patterns in total iron and manganese concentrations in a reservoir tailwater (East Sidney Lake, NY). (Hydrograph provided for reference.)

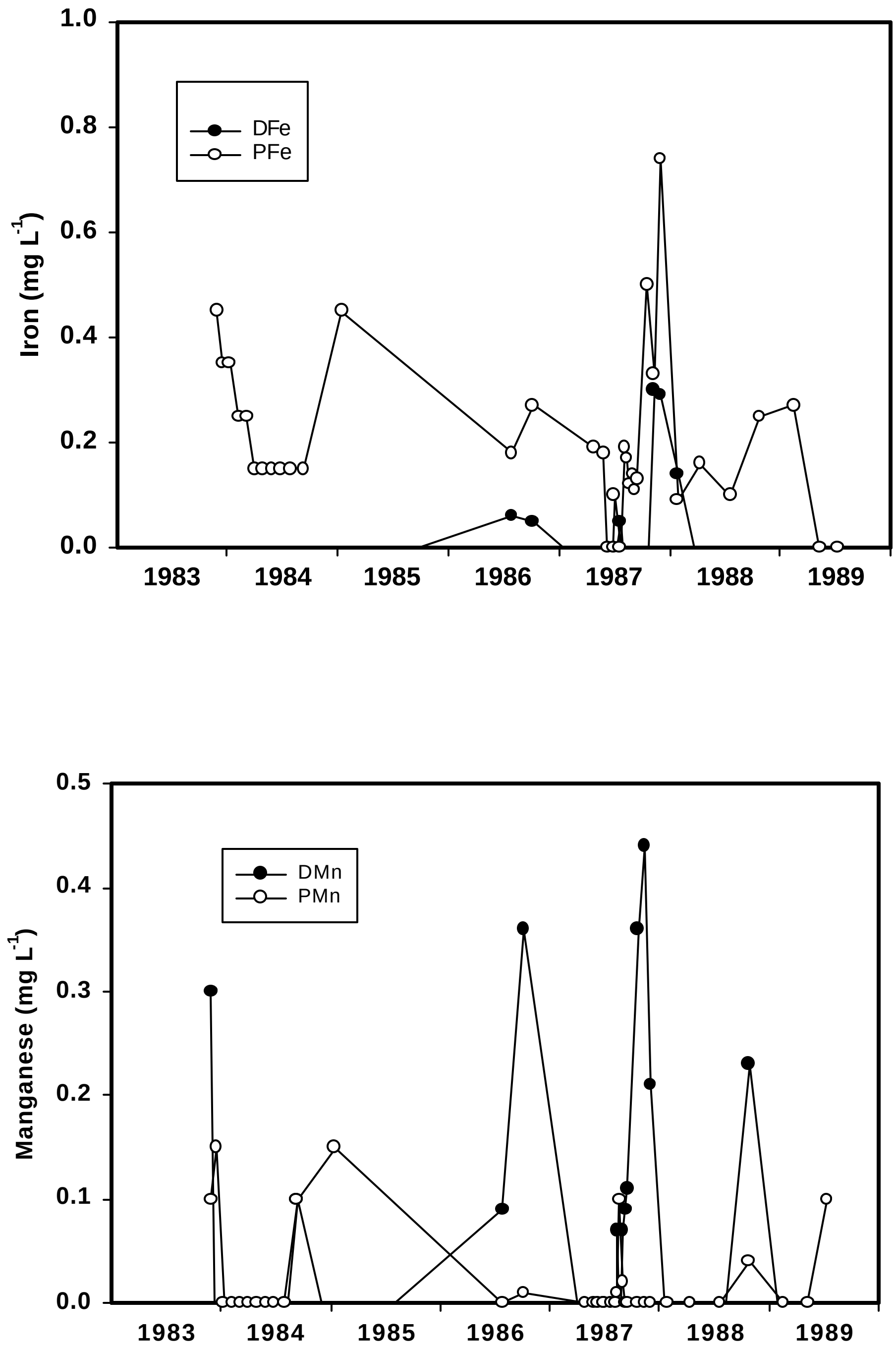


Figure 1.3.17 Temporal patterns in dissolved and particulate iron and manganese concentrations in a reservoirs tailwater (Lake Hartwell, GA/SC)

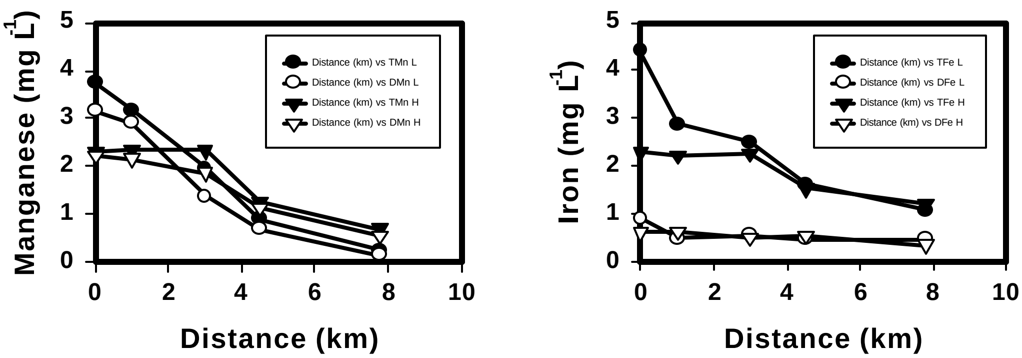


Figure 1.3.18 Spatial trends in total and dissolved iron and manganese concentrations in reservoir tailwaters during high (H) and low (L) flow (Nimrod Lake, AR).

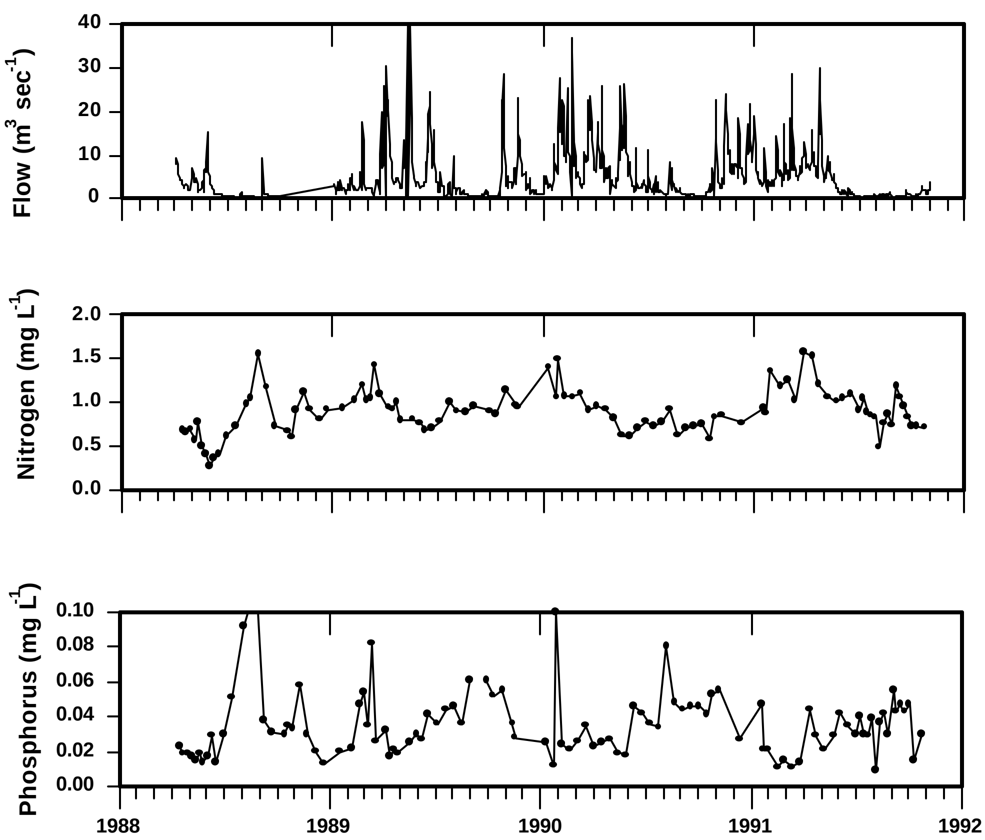


Figure 1.3.19 Temporal patterns of total nitrogen and phosphorus concentrations in reservoir tailwaters (East Sidney Lake, NY).
(Hydrograph provided for reference.)

be expected while higher nitrate concentrations would be expected during oxic periods. Distribution of total phosphorus concentrations during the summer growing season indicate maximum values occur in late summer, coincident with anoxia maxima in the upstream impoundment (Figure 1.3.19). In general, during thermal stratification and increased concentrations in the releases, ammonia nitrogen concentrations decrease with distance from dam and nitrate concentrations increase with distance from dam (Figure 1.3.20). Observations of concentrations at a low and high flow indicate that ammonia concentrations remaining higher for a greater distance during high flow before decreasing. Nitrate concentrations increase with distance and the rate of increase is greater during the low flow. Possible mechanisms for these trends include biological activity and physicochemical processes that vary with flow.

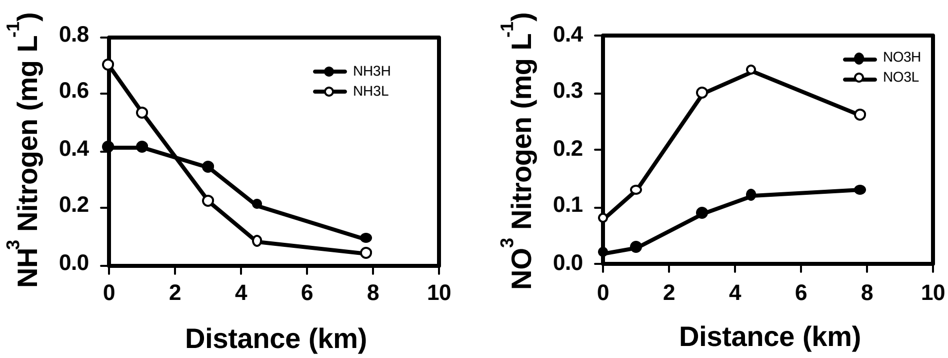


Figure 1.3.20 Spatial trends in ammonia and nitrate concentrations in reservoir tailwaters during high (H) and low (L) flow (Nimrod Lake, AR).

Particulate material, provided to the river from runoff and resuspension, impacts river water quality by providing sites for the adsorption and desorption of chemical constituents. Particulates may also be relatively inert and concentrations of suspended material will change as a function of grain size and flow. Major particulate substances include clay minerals, weathered minerals, and organic matter (carbon) such as leaves and twigs (pieces to logs). In general, particulate matter concentrations increase with discharge to the impoundment. The increases can be substantial, e.g. 572 and 1440 times above minimum discharge for particulate organic matter have been reported (Perry and Perry 1991). However, the impoundment buffers the transport of particulate material and suspended solids in the tailwater are typically lower and less variable than upstream values (Figure 1.3.21).

In most rivers and streams (of which tailwaters are a subset) the turbulent mixing ensures a relatively uniform distribution of dissolved substances, although discontinuities may exist especially at confluences, near-bank boundaries, and in deep rivers. Turbulence usually results in an equilibrium between dissolved gases and the atmosphere. Exceptions to this occur in areas of high productivity and reservoir tailwaters where supersaturation occurs. Major dissolved substances follow the "bicarbonate system" described for reservoirs including relationships of calcium and magnesium from weathering. Longitudinal gradients exist with concentrations decreasing with distance from the source. Nutrients are

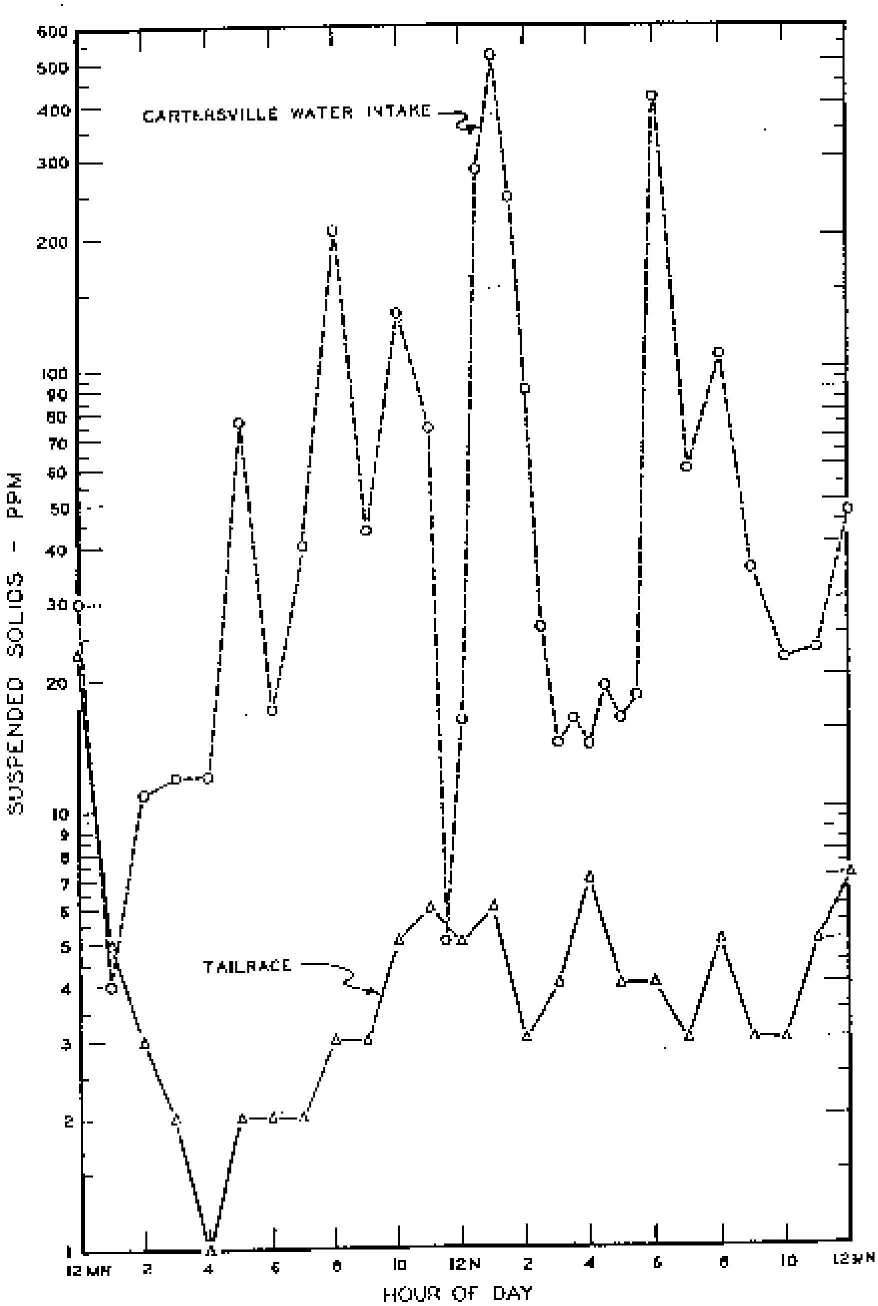


Figure 1.3.21 Variation of suspended solids with time in the tailwater of Allatoona Lake, GA (note relatively low and constant concentrations) and at the Cartersville water intake (note increased variability more typical of riverine conditions (Krenkel 1965) (From Kenkel and Novotny (1980), reprinted with permission of Academic Press

primarily available from the upstream impoundment, agricultural or other areas in the adjacent watershed, and point sources. Carbon is also present in soluble forms through leaching and breakdown of particulate carbon.

1.3.7 BIOLOGICAL PROCESSES

Tailwaters are more than conduits for transporting particulate and dissolved materials. Biological processes occur in flowing waters and influence the cycling of transported material. Running water is a richer habitat than still water due to currents which prevent the accumulation of a shell of depleted resources around organisms by providing a constant supply of fresh material (nutrients, oxygen) for metabolism. Biological processes in tailwaters involve benthic communities (microbial, invertebrate, and vertebrate), macrophytes, periphyton, phytoplankton, zooplankton, and fisheries communities. Other biota (birds, reptiles, amphibians, etc.) are usually considered on a site-specific basis.

Biological processes are often inseparable from physical and chemical process as is illustrated in evaluating the dynamics of dissolved oxygen in tailwaters. The consumption of dissolved oxygen in rivers is most often a biological process which breaks down organic matter (carbon). This process often results in a decrease (sag) in dissolved oxygen concentrations downstream from the input of organic matter. As illustrated in Figure 1.3.22, the "sag" in dissolved oxygen concentrations can be

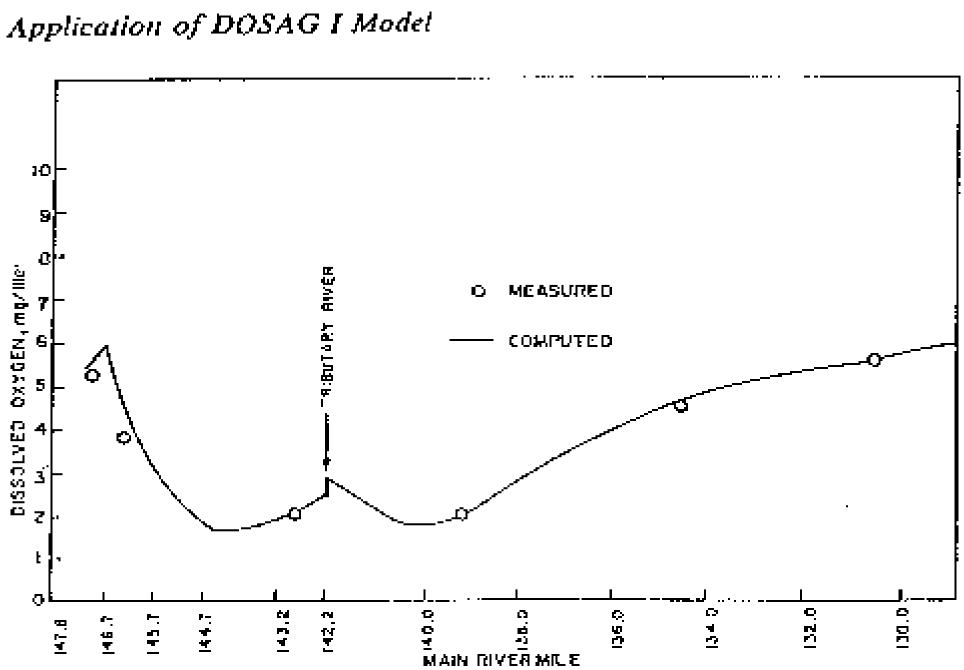


Figure 1.3.22 Measured and computed dissolved oxygen concentrations demonstrating “sags” in longitudinal concentrations. (From Krenkel and Novotny (1980), reprinted with permission of Academic Press).

predicted. Reaeration is the physical process by which turbulence at the air water interface provides dissolved oxygen to the river through entrainment of air at the air/water interface. Since these processes are related, they may be summarized using an equation (or a derivation) referred to as the Streeter-Phelps equation (Streeter and Phelps 1925):

$$D = \frac{K_1 L_O}{K_2 - K_1}\left(e^{-K_1 t} - e^{-K_2 t}\right) + D_O e^{-K_2 t}$$

where, D = the oxygen deficit, L_0 is the ultimate BOD, K_1 is the deoxygenation coefficient, K_2 is the reaeration coefficient, t is the time, and D_0 is the oxygen deficit. The Streeter-Phelps equation has been modified numerous times to include other sources and sinks of dissolved oxygen (i.e. sedimentation, additional inputs, biological processes such as photosynthesis and respiration). Nitrogen also exerts a demand on dissolved oxygen but at a much slower rate than carbon (about 20 days are required for bacterial acclimation) and is a second order reaction in variations of the above equation. This type of approach is useful in evaluating the recovery distance for dissolved oxygen concentrations at various discharges.

The removal of reduced metals in tailwaters is also a function of physical, chemical, and biological processes. While often modeled using chemical oxidation kinetics, Dortch et al. (1992) recognize the significance of biological removal processes and association with streambed substrate. Biological processes of metals removal in tailwaters have not been widely studied but inferences from other freshwater and marine systems can be made. Good summaries of biological mechanisms for metals removals are provided in Chapnick et al. (1982), Nealson et al. (1989), and Ghiorse (1984).

The effects of reservoir releases on invertebrates and fisheries and on tailwater ecology have been summarized in literature reviews prepared by Walburg et al. (1980 and 1981) and discussed by Petts (1984). Field studies conducted at seven reservoirs (Walburg et al. 1983) and fish recruitment and movement in a flood control reservoir (Jacobs et al. 1985) suggest a variety of impacts on the biotic community associated with reservoir operations. Detailed summaries of project impacts are presented in Nestler et al. (1986) and comparisons to pre-project conditions are also included. In general, invertebrates (particularly Ephemeroptera) responded to varied flow conditions with a tendency to increase in number with distance from the dam. For surface release projects, the tailwater benthos was dominated by filter-feeding organisms, such as net-building caddisflies. Deep-release or bottom withdrawal projects, with anaerobic hypolimnia providing a clear, nutrient-rich release foster periphyton development in the tailwater. Deep-release projects with aerobic hypolimnia provided a source for phytoplankton, zooplankton, and fishes to the tailwater. The benthos in these tailwaters were often dominated by grazers, such as some species of chironomids, oligochaetes, amphipods, and isopods. The "surge" associated with hydropower production redistributes macrophytes, periphyton, macroinvertebrates, and even fishes in a downstream direction. Fluctuation of flow (hence wetted perimeters) is also a major adverse impact to the benthic community through dislodgement (Anderson and Lehmkuhl 1968), stranding (Trotzky and Gregory 1974), or overall changes to daily cycles (Perry and Perry 1986). However, proper management of flow modification at hydropower facilities may protect and enhance the community downstream (Morgan et al. 1991).

Studies conducted by Blinn et al. (1989) with diatoms from the tailwater of Glen Canyon Dam, a hydropower project on the Colorado River in Arizona, demonstrated temperature effects on community structure with a threshold between 12 and 18 ^{O}C. They further suggest that changes in diatom community structure, associated with changes in temperatures as a result of a change in operations, could impact higher trophic levels such as macroinvertebrate grazers. Angradi and Kubly (1993) suggest that fluctuation in discharge and resultant exposure to the atmosphere downstream from Glen Canyon Dam, lowers the resistance of the epiphytic community, decreases the rate of recolonization, and decreases gross primary production. A longer term study of the Upper Rhône River, France (Dolédec, et al. 1996) describes a pre-project benthic community dominated by rheophilic species (Trichoptera, hydropsychids such as caddisflies) which changed to a community with mostly lentic and thermophilic species (e.g., flatworms, gastropods, leeches, crustaceans) after impoundment. Changes were observed at all sites (both upstream and downstream from a diversion dam, by-passed section, and power station) and were attributed to a progressive warming of the water and newly created habitat.

Petts (1984) provides a good discussion of the ecological impacts of reservoirs in relation to effects on life cycles (hatching, growth, and emergence) of invertebrates and how changes to the invertebrate community can effect the fisheries. Cues to life stages, such as temperature requirements, water level, and substrate types) are altered with the construction of the impoundment and community structure of the tailwater can change dramatically. Ecologically, changing the macroinvertebrate community can impact the fisheries and result in a very different biological community compared to the “pre-reservoir” community.

1.3.8 SUMMARY

Reservoir tailwaters, while exhibiting water quality processes observed in unregulated rivers, differ due to water quality influences from the upstream impoundment and varied flow due to operation of the dam. Both positive and negative impacts can be associated with the impoundment of a river and effect in the downstream area or tailwater. Habitats can be either degraded or improved, water quality conditions can decline or be enhanced spatially and temporally (compared to inflow conditions). Physical, chemical, and biological processes are often inseparable resulting in the need for an ecosystem approach to the management of tailwaters that may often be limited by project operations.

Table 1.3.1 Calculated Mean Velocities (cm sec^{-1}) for Various Slopes and Morphometries			
Slope (m km^{-1})	200 m wide 4 m deep	20 m wide 0.5 m deep	2 m wide 0.25 m deep
0.5	150	50	25
1.0	(250)*	60	35
2.0		80	50
5.0		130	70
10.0		180	100
*This figure is above the normal threshold for enlarged channels.			

Table 1.3.2. Current Velocity and Substrate Size			
Velocity Range		General Bottom Composition	App. Diameter (mm)
(cm sec^{-1})	(ft sec^{-1})		
1-20	0.1-0.7	silt, mud, (organic debris)	<0.02
20-40	0.7-1.3	fine sand	0.1-0.3
40-60	1.3-2.0	coarse sand to fine gravel	0.5-8
60 -120	2.0-3.9	small, med to large gravel	8-64
120-200	3.9-6.6	large cobbles to boulders	>128

Tables 1.3.1 and 1.3.2 were modified from Einsele (1960) Die Stromungsgeschwindigkeit als beherrschender Fakto bei der limnologischen Gestaltung der Gewasser, *Osterreichs Fischerei* 2:1-40.

1.3.9 REFERENCES

Anderson, N. H. and Lehmkuhl, D. M. 1968. "Catastrophic drift of insects in a woodland stream," *Ecology*, 49:198-206.

Angradi, T. R. and Kubly, D. M. 1993. "Effects of atmospheric exposure on chlorophyll *a*, biomass and productivity of the epilithon of a tailwater river," *Regulated Rivers: Research and Management*, 8:345-358.

Ashby, S. L., Kennedy, R. H., and Jabour, W. E. 1995. ‘Water quality dynamics in the discharge of a southeastern hydropower reservoir: response to peaking generation operation.” *Lake and Reservoir. Manage*., 11(3):209-215.

Barillier, A., Garnier, J., and Coste, M. 1993. “Experimental reservoir water quality release: impact on the water quality on a river 60 km downstream (Upper Seine River, France).” *Wat. Res.*, 27(4):635-643.

Blinn, D. W., Truiit, R., and Pickart, A. 1989. “Response of epiphytic diatom communities from the tailwaters of Glen Canyon Dam, Arizona, to elevated water temperature,” *Regulated Rivers: Research and Management*, 4:91-96.

Chapnick, S. D., Moore, W. S., and Nealson, K. H. 1982. “Microbially mediated manganese oxidation in a freshwater lake.” *Limnology and Oceanography* 27(6):1004-1014.

Cole, G. A.. 1983. *Textbook of Limnology*, 3rd Ed., The C.V. Mosby Company, St. Louis.

Dawley, E. M. 1986. “Effects of 1985-86 levels of dissolved gas on salmonids in the Columbia River.” Seattle, WA: National Marine Fisheries Service.

Daniil, E. I., Gulliver, J., and Thene, J. R. 1991. “Water-quality impact assessment for hydropower,” *J. Environ. Eng*., 117(2):179-193.

Dolédec, S., Dessaix, J., and Tachet, H. 1996. “Changes within the Upper Rhône River macrobenthic communities after the completion of three hydroelectric schemes: anthropogenic effects or natural change?,” *Arch. Hydrobiol*., 136(1):19-40.

Dortch, M. S., Tillman, D. H., and Bunch, B. W. 1992. “Modeling water quality of reservoir tailwaters,” Technical Report W-92-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Ebel, W. J., Raymond, H. L., Monan, G. E., Farr, W. E., and Tanonaka, G. K. 1975. “Effects of atmospheric gas supersaturation caused by dams on salmon and steelhead trout of the Snake and Columbia Rivers.” Seattle, WA: National Marine Fisheries Service.

Ghiorse, W. C. 1984. “Bacterial transformations of manganese in wetland environments.,” *Current Perspectives in Microbial Ecology, Proceedings of the Third International Symposium on Microbial Ecology*, Klug, M.J. and Reddy, C.A., eds., American Society for Microbiology, Washington, D.C. pp. 615-622.

Hudgins, M. H. 1995. "Water quality changes as a result of coldwater releases in Lake Moomaw," in Water Quality '94, Proceedings of the 10th Seminar, 15-18 February 1994, Savannah, GA, Miscellaneous Paper W-95-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Hynes, H. B. N. 1970. *The Ecology of Running Waters*, University of Toronto Press, Toronto, Canada.

Jabour, W. E., Satterfield, J. M., Hains, J. J., and Carroll, J. H. 1996. "Late-summer temperature and dissolved oxygen dynamics in the tailrace of a southeastern hydropower reservoir, " in Water Quality '96, Proceedings of the 11th Seminar, 26 February-01 March 1996, Seattle, WA, Miscellaneous Paper W-96-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Jacobs, K. E., Swink, W. D., Nestler, J. M., and Curtis, L.T. 1985. "Fish Recruitment and Movement in a Flood Control Reservoir and Tailwater," Technical Report E-85-15, prepared by U.S. Fish and Wildlife Service and Environmental Laboratory for U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Krenkel, P.A. 1965. Investigation of manganese concentration in Allatoona Reservoir, Georgia. Rep. To U.S. Army Corps of Eng., Mobile District, Alabama.

Krenkel, P. A. and Novotny, V. 1980. *Water quality management*, Academic Press, New York.

Morgan, II, R. P., Jacobsen, R. E., Weisberg, S. B., and McDowell, L. A. 1991. "Effects of flow alteration on benthic macroinvertebrate communities below the Brighton hydroelectric dam," *J. Freshwat. Ecol.*, 6(4):419-429.

Nealson, K. H., Tebo, B. M., and Rosson, R. A. 1989. "Occurrence and mechanisms of microbial oxidation of manganese." *Advances in Applied Microbiology* 33:279-318.

Neel, J. K. 1951. "Interrelations of certain physical and chemical features in a head-water limestone stream." *Ecology*, 32:368-391.

Nestler, J. M., Walburg, C. H., Novotny, J. F., Jacobs, K. E., and Swink, W. D. 1986. "Handbook on Reservoir Releases for Fisheries and Environmental Quality," Instruction Report E-86-3, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Nix, J., Hamlin-Tillman, D. E., Ashby, S. L., and Dortch, M. S. 1991. "Water quality of selected tailwaters." Technical Report W-91-2. US Army Engineer Waterways Experiment Station, Vicksburg, MS.

O’Connell, R.L., Geckler, J.R., Clark, R.M., Cohen, J.B., and Hirth, C.R. 1962. Report of survey of the Truckee River. U.S. Dept. of Health, Educ. and Welfare, Public Health Serv., Cincinnati, Ohio (July).

Palmer, R. W. and O’Keefe, J. H. 1990. “Downstream effects of impoundments on the water chemistry of the Buffalo River (Eastern Cape), South Africa,” *Hydrobiologia*, 202:71-83.

Perry, S. A. and Perry, W. B. 1986. “Effects of experimental flow regulation on invertebrate drift and stranding in the Flathead and Kootenai Rivers, Montana, USA,” *Hydrobiologia*, 134:171-182.

Perry, S. A. and Perry, W. B. 1991. “Organic carbon dynamics in two regulated rivers in northwestern Montana, USA,” *Hydrobiologia*, 218:193-203.

Petts, G. E. 1984. *Impounded rivers, perspectives for ecological management*, John Wiley and Sons, NY.

Sear, D. A. 1995. “Morphological and sedimentological changes in a gravel-bed river following 12 years of flow regulation for hydropower,” *Regulated Rivers and Management*, 10:247-264.

Streeter, H. W. and Phelps, E. B. 1925. “A study of the pollution and natural purification of the Ohio River,” *Public Health Bull*., 146, U.S. Public Health Serv., Washington, D.C.

Trotzky, H. M. and Gregory, R. W. 1974. “The effects of water flow manipulation below a hydroelectric power dam on the bottom fauna of the upper Kennebec River, Maine,” *Trans. Amer. Fish. Soc*., 103:318-324.

U. S. Army Corps of Engineers. 1996. “Water Quality ‘96, Proceedings of the 11th Seminar, 26 February-01 March 1996, Seattle, WA,” Miscellaneous Paper W-96-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Vannote, R. L., Minshall, G. W., Cummins, K. E., Sedell, J. R., and Cushing, C. E. 1980. “The river continuum concept,” *Can. J. Fish. Aquat. Sci*., 37:130-137.

Walburg, C. H., Novotny, J. F., Jacobs, K. E., Campbell, T. M., and Swink, W. D. 1980. "Water Quality, Macroinvertebrates, and Fisheries in Tailwaters and Related Streams; An Annotated Bibliography," Technical Report E-81-8, prepared by the U.S. Department of the Interior, for the U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss.

Walburg, C. H., Novotny, J. F., Jacobs, K. E., Swink, W. D., Campbell, T. M., Nestler, J., and Saul, G. E. 1981. "Effects of Reservoir Releases on Tailwater Ecology: A Literature Review," Technical Report E-81-12, prepared by U.S. Department of the Interior, Fish and Wildlife Service, National

Reservoir Research Program, East Central Reservoir Investigations, and Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, for the U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss.

Walburg, C. H., Novotny, J. F., Jacobs, K. E., and Swink, W. D. 1983. "Effects of Reservoir Releases on Water Quality, Macroinvertebrates, and Fish in Tailwaters: Field Study Results," Technical Report E-83-6, prepared by National Reservoir Research Program, U.S. Fish and Wildlife Service, for the U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Miss.

Ward, J. V. and Stanford, J. A. 1983. "The serial discontinuity concept of lotic ecosystems," in T.D. Fontaine and S.M. Bartell (eds), *Dynamics of lotic ecosystems*, Ann Arbor Science Publishers.

CHAPTER 2

PROBLEM IDENTIFICATION AND ASSESSMENT

INTRODUCTION

Perhaps one of the most important aspect of water quality enhancement is identification of the water quality problem(s) and a robust analysis of the cause(s). This is the next step after an understanding of watershed-reservoir processes has been achieved. Problem identification requires detection of the problem, definition of the problem (e.g., what is the problem as perceived by water resource users or indicators), and identification of possible causes or sources of the problem. Often, a specific and detailed assessment of processes in the watershed, reservoir, and tailwater are necessary to develop the level of understanding in a particular system so that feasibility of enhancement techniques can be evaluated. This session will provide an overview of problem definition with input from workshop participants and detailed information for assessing processes in the watershed, reservoir, and tailwater.

2.1 WATER QUALITY PROBLEM DEFINITION AND DETECTION

What is a water quality problem? This answer to this question is a function of many factors such as: user needs, user perceptions, regional conditions, and water quality indicators.
A water quality problem is often not considered problematic until a user is no longer able to use the resource due to water quality. For example, an algal bloom that decreases turbidity (water clarity) at a swimming beach such that the beach must be closed becomes a water quality problem. This same algal bloom may or may not impair other uses such as fishing, boating, or water supply. Consequently, the perception of a problem will be different depending on the user. Furthermore, in an area of the country with highly productive lakes and reservoirs, algal blooms are more common than in areas of low productivity and may be more tolerable. The timing and magnitude of the algal bloom may also determine the extent of the problem. The use of water quality indicators may help in evaluating water quality problems but are most appropriately applied on a regional basis with consideration for user needs.

2.1.1 PERCEPTIONS OF WATER QUALITY PROBLEMS

Beauty is in the eye of the beholder and water quality problems are often defined by perceptions based on user experiences and requirements. A person accustomed to a eutrophic system with high algal biomass and low water clarity may be satisfied with the local conditions until he or she travels to an area with considerably better water quality. Upon return to the local spot, it becomes apparent that water quality is substandard and has been all along. Fishermen have different water quality perceptions than do recreational boaters suggesting user requirements may influence water quality perceptions. Just as important, are perceptions of water quality by nonusers. For example, a farmer who has cows with access to a reservoir inflow may not ever fish or swim in the lake downstream and, therefore, has no real opinion about the local water quality. If swimming is impaired, due to decreased visibility, the connection between the algal bloom and the farmer's cows may not be easily conveyed to the farmer.

2.1.2 WATER QUALITY BIOCRITERIA AND INDICATORS

Relating actual water quality data to user perceptions may be a reasonable approach defining water quality problems and developing enhancement techniques. Heiskary and Walker (1988) developed a method for establishing phosphorus criteria in Minnesota Lakes that took traditional predictive techniques for phosphorus, chlorophyll, and water clarity for establishing criteria and calibrated the method to include user perceptions. This approach suggests that criteria be applicable to user perceptions and regional differences in water quality.

More recently, the EPA and others in the scientific community, developed an approach for conducting a bioassessment of lakes and reservoirs and establishing biocriteria (USEPA 1998). In this approach, bioassessment is defined as an evaluation of the biological condition of a waterbody that uses biological surveys and other direct measurements of resident biota in surface waters. Biological criteria (biocriteria) are numeric values or narrative expressions that describe the reference biological condition of aquatic communities inhabiting waters of a given designated aquatic life use (USEPA 1998).

Once a biological survey has been conducted and biocriteria are established the information can be used for:

- Problem screening and identification
- Assessing the effectiveness of implemented water resource management practices
- Determining attainment of designated aquatic life uses
- Refining aquatic life use categories
- Identifying impact sources.

Clearly, collaboration among users and resource agencies is required in this process. Much of the data used in determining biocriteria come from the state's monitoring programs, federal data collection efforts, and studies/monitoring conducted by academic institutions and the private sector. Additional information on biocriteria is available in USEPA 1990, 1991, 1992, 1996a, 1996b, and 1996c.

Indicators of water quality problems may be defined by paraphrasing a definition by the Intergovernmental Task Force on Monitoring Water Quality (ITFM 1995, Appendix E) which defines environmental indicators as, "... measurable feature or features that provide managerially and scientifically useful evidence of environmental and ecosystem quality or reliable evidence of trends in quality." Characteristics of indicators include technical considerations such as validity, sensitivity, and representativeness (i.e., one indicator for a whole system may not be appropriate), practical considerations such as cost and logistical difficulty, and programmatic considerations such as understandability and relevance. Indicators may be classified as those that provide information for assessment (compliance documentation), trend detection (documenting change, both positive and negative), early warning (anticipate future conditions), or diagnosis (identify the causative agent) (Cairns et al. 1993). Indicators may be developed from morphometric features, physicochemical constituents, specific nutrient and chemical concentrations, biota, and integrative values (Table 2.1).

An earlier definition (Hunsaker and Carpenter, 1990) suggests indicators may be defined as "... a characteristic of the environment that, when measured, quantifies the magnitude of stress, habitat characteristics, degree of exposure to the stressor, of degree of ecological response to the exposure." This definition seems applicable to 18 indicators listed in the U.S. Environmental Protection Agency's "18 Environmental Indicators of Water Quality" which allows for an approach that detects "problems" that would not necessarily be detected with traditional compliance monitoring. These indicators are good, as a first cut laundry list of a total range of concerns. The first 4 are for water supply engineers/public health officials. Metrics for these indicators may include variables not measured by lake professionals. These 18 indicators are:

1. Population served by drinking water systems violating health-based requirements
2. Population served by unfiltered surface water systems at risk from microbiological contamination
3. Population served by community drinking water systems exceeding lead action levels
4. Drinking water systems with source water protection programs
5. Fish consumption advisories
6. Shellfish-growing waters approved for harvest for human consumption
7. Biological integrity of rivers and estuaries
8. Species at risk of extinction
9. Rate of wetland acreage loss
10. Designated uses: drinking water supply, fish and shellfish consumption, recreation, aquatic life
11. Groundwater pollutants (nitrate)
12. Surface water pollutants
13. Selected coastal surface water pollutants in shellfish
14. Estuarine eutrophication conditions
15. Contaminated sediments
16. Selected point source loadings to surface water and groundwater
17. Nonpoint source sediment loadings from crop land
18. Marine debris

Table 2.1 Features Useful for Developing Indicators	
Feature	Description
Morphometry	Lake or reservoir dimensions Extent of stratification and mixing (Froude number, lake number)
Physicochemical Constituent	Dissolved Oxygen - Balance between production and consumption; Oxygen status (profile, deficit rate, anoxic factor) Water clarity - Light attenuation (color, inorganic particles, organic particles (algae))
Nutrient and Chemical Values	Ambient concentrations (compliance, toxicity, eutrophication status (8.0, 26.7, 84.4 μg L^{-1} Total Phosphorus and 661, 753, 1875 μg L^{-1} Total Nitrogen (geometric means) for oligotrophic, mesotrophic, and eutrophic systems, respectively (OECD, 1982)) Loading rates and permissible limits (Total Maximum Daily Load)
Biological Components	Indicator species (pollution tolerant/intolerant) - pitfalls include ambiguity and relation to community structure/function Biotic diversity (e.g., Shannon and Weaver, 1949; Wilhm, 1967) Morphoedaphic Index - Fish harvest (Ryder, 1965)
Integrative Components	Trophic state Trophic State Index (TSI, Carlson, 1977) Index of Biotic Integrity (IBI, Karr et al., 1986) EPT Index - based on the abundance of selected orders of pollution-sensitive insect larvae (Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)). EPT Index is the sum of the number of families within these orders.

While there is obvious overlap in these two lists of indicators, using concepts from both may enable identification of specific problem sources (via compliance monitoring) and an overall assessment of environmental health (cumulative effects even within established standards and criteria). Recent discussions suggesting water quality standards for metals toxicity may need reconsidering (due to site specific and metal specific responses to water quality) (Renner, 1997) are also indicative of a need to carefully consider the development and application of indicators.

Using perceptions of water quality problems, as defined by users and quantification of problems using indicators applied on a regional basis, is a good start in problem identification. An example of a program that utilizes a similar approach is the Tennessee Valley Authority's (TVA) Assessment Program. Specific techniques for using water quality indicators have also been developed by the U.S. Department of Agriculture Soil Conservation Service (Terrell and Perfietti, 1989).

2.1.3 PROBLEM SOURCES

Problem identification includes consideration of the source(s) of the problem so that application of enhancement techniques can be targeted for maximum benefit. Often more than one source exists.

For example, multiple nonpoint sources may be contributing to eutrophication related problems in the downstream reservoir and, if this has been an ongoing situation, the reservoir sediments may also be contributing significantly to the problem. Assessment of processes in the watershed, reservoir, and tailwater, as previously discussed, is absolutely necessary in identifying the source(s) of the problem. These assessments can lead to some surprising results. In a recent study of a large multi-river watershed in the southeast, the Apalachicola-Chattahoochee-Flint River Basin, urban and suburban land use accounted for only 5% of the basin but was considered to have the most important effect on stream-water quality (U.S. Geological Survey, 1998).

Other good sources for assessing water quality using indicators include bioassessment protocols (Plafkin et al., 1989) and websites for decision support systems such as WATERSHEDDS (Osmond et al., 1997). The URL for this website is **http://h2osparc.wq.ncsu.edu.**

2.1.4 OVERVIEW OF CONTAMINANTS, SPILLS, ETC.

Introduction. Chemicals associated with discharges from industry, agricultural, and mining practices, and chemical spills are often considered as contaminants instead of being included in eutrophication problems previously described. Contaminants may be grouped as heavy metals (e.g., cadmium, lead, zinc), specific elements of concern such as mercury, arsenic, and selenium, hydrocarbons such as polychlorinated biphenyls (PCBs), an array of pesticides and herbicides, and others. In general, these contaminants respond to water quality conditions such as anoxia and pH, may be biologically magnified (increase in concentration in organisms from the lower to the upper levels of the food chain), and remain in the environment for a wide range of time. These problems are often site or event specific and are not subject to conventional enhancement techniques mentioned below. However, a brief overview is provided with the recommendation for more detailed review of the literature for those with greater interests.

2.1.4.1 HEAVY METALS

Most often, these contaminants are partitioned into a soluble phase and a particulate phase associated primarily with clay minerals which are easily transported as suspended material (Figure 2.1.1). Cadmium, iron, and zinc are associated with particulate transport as colloids. Heavy metals are also associated with lower molecular weight fractions of organic carbon and labile organic complexes which are more electrochemically active and therefore better sites for metal sorption. Bacteria and inorganic materials often provide the surfaces required for adsorption. Metal sorption is a function of pH (sorption decreases as pH decreases, Figure 2.1.2), suspended solids concentrations (sorption increases as suspended solids increase due to an increase in site availability), and particle composition (sorption increases as organic surface area increases). Metals and non-labile organic compounds adsorb to particles and desorb as pH decreases. Surface charge of particles is pH dependent - at low pH a positively charged surface prevails at high pH a negatively charged surface prevails. Under normal surface water pH - silica, clays, feldspars, and manganese oxides are negatively charged and have a strong affinity for positively charged ions.

2.1.4.2 MERCURY

Mercury has received much attention lately, particularly since concentration increases up the food chain (biomagnification). Mercury enters an aquatic system primarily from agricultural fungicides, mining and smelting operations, industrial discharge, atmospheric input, flooded soils, and via mineral weathering processes. Mercury can occur in concentrations that are toxic (0.03 (*Scenedesmus*) to 3.0 ppm for snails).

2.1.4.3 POLYCHLORINATED BIPHENYLS (PCBS)

PCBs are found as ingredients in lubricants, hydraulic fluids, transformer fluids, asphalt, and waxes and exist in 210 possible forms. Generally, PCBs are long-lived in the environment, associated with very fine sediments (clays) and while not readily mobilized by chemical reactions, they can move up the food chain through biomagnification. The relationship of PCB contaminated sediments to macroinvertebrate and fish communities have been evaluated for Lake Hartwell, a CE project on the Georgia and South Carolina border (Alexander 1995). Study results indicate biomagnification, adverse impacts to fish health and vitality, and a persistent problem downstream of the contaminated site.

2.1.4.4 PESTICIDES/HERBICIDES

These contaminants are chemical compounds that vary greatly in how long they remain in the environment as problematic compounds. Chlorinated hydrocarbons (organochlorines, e.g., DDT, dioxin, chlordane, myrex, aldrine, etc.) are very persistent. Organophosphorus compounds (parathion, malathion, diazanon, etc.) are not very persistent and often photosensitive resulting in a limited time for measurable concentrations to exist in the aquatic system. Carbamate compounds (e.g., carbaryl and Seven) are not very persistent and are intended to kill pests mostly by working on the nervous system. Others include inorganic pesticides that contain mercury, arsenic, and lead. Recently, lethal compounds have been restricted and except for misuse, most currently used pesticides are short-lived in aquatic systems. The most notable exception is DDT and its derivatives which are long-lived in sediments and move up the food chain.

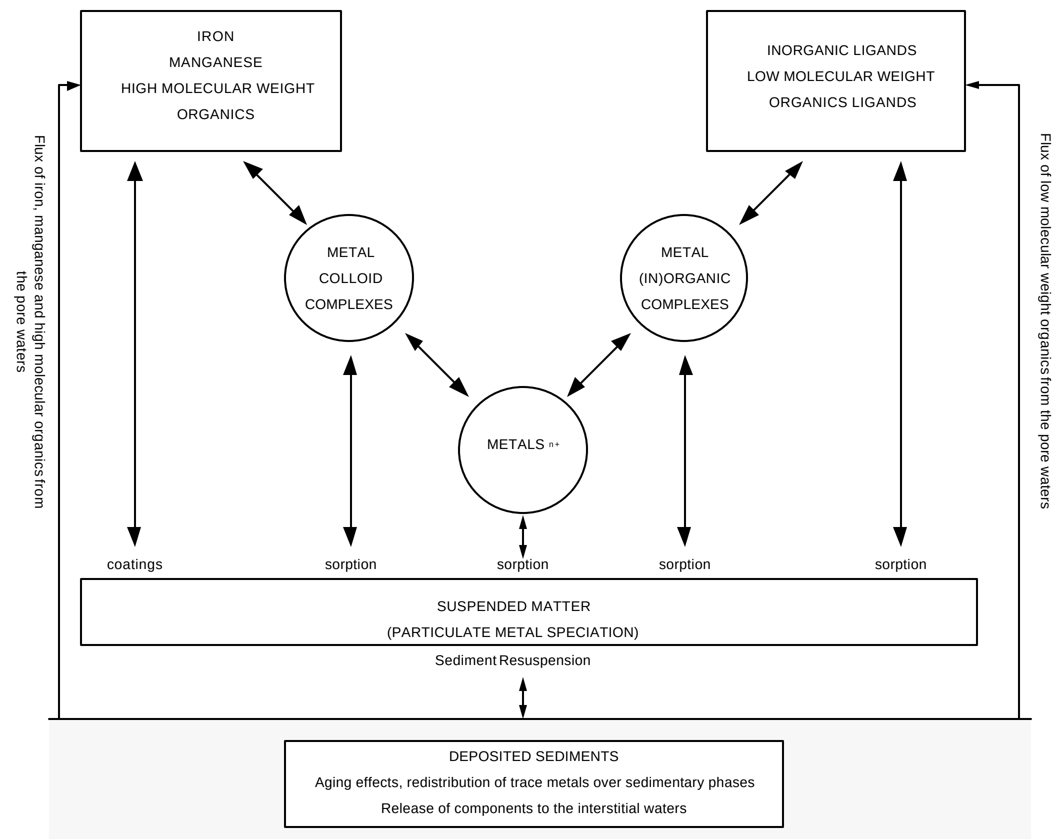
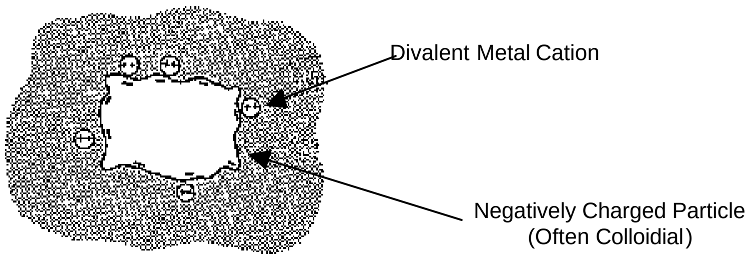


Figure 2.1.1 Summary of major processes and mechanisms in the interactions between dissolved and solid species in surface waters.



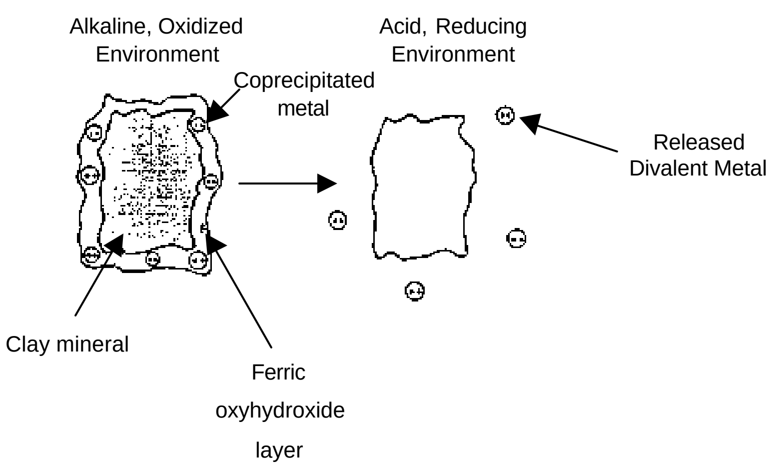


Figure 2.1.2 Dissolution of ferric oxyhydroxide layer and release of coprecipitated metals in an acid, reducing environment

2.1.4.5 CHEMICAL SPILLS

Contaminants entering an aquatic system via a spill are often associated accidents occurring during transportation of the chemical, inadvertent spills associated with agricultural and industrial operations, or improper disposal. Known incidents of chemical spills can be monitored for movement within and impacts to the aquatic system. If the spilled chemical is hydrophobic, it will often remain relatively unmixed with the aquatic system allowing easier tracking and cleanup.

Spillage or leakage of gasoline (and additives) associated with boats and refueling stations is also a source of contaminants to the water and sediment. Yagecic and Moy (1995) have demonstrated that lead concentrations in sediments are greater near refueling points.

2.1.4.6 CONTAMINANT DETECTION

The detection of contaminants in an aquatic system is usually through one of three ways, 1) direct observation such as in oil spills, 2) indirect evidence such as a fish kill, an unusually large number of physical deformities in biota, or a high incidence of illness in the local population, or 3) chemical analyses ranging from screening levels to extensive surveys. The latter of these methods are often the result of one of the first two since, typically, the analytical costs for the analysis of most of these methods are often quite high. In addition to sample collection for analysis, in situ techniques such as fluorescence (Cremeans et al. 1995) and remote sensing techniques such as aerial/satellite detection methods may be used to monitor the transport of spills.

2.1.4.7 CONTAMINANT MANAGEMENT AND REMEDIATION

Often techniques used to manage for water quality problems associated with eutrophication are applicable to management of contaminants discussed in this section. For example, heavy metals mobilized by low or undetectable concentrations of dissolved oxygen may be successfully managed with aeration devices. Methods, such as chemical additions to alter pH may be applicable for decreasing contaminant mobility. A "no action" alternative which allows contaminated sediments to be buried with time (for sediments subjected to minimal disturbances or redistribution, e.g., sediments in accumulation zones (Figure 2.1.3)) may be feasible. This method works for contaminants that remain bound to the sediments. Other alternatives may involve operational techniques, such as manipulation of residence time and dilution, and specific clean up methods used for spills. For contaminants in aquatic sediments that are primarily cycled via biotic mechanisms, separation of the contaminated sediments from the biota using capping, removal (dredging), or isolation (dikes, etc.), is a possible means of management.

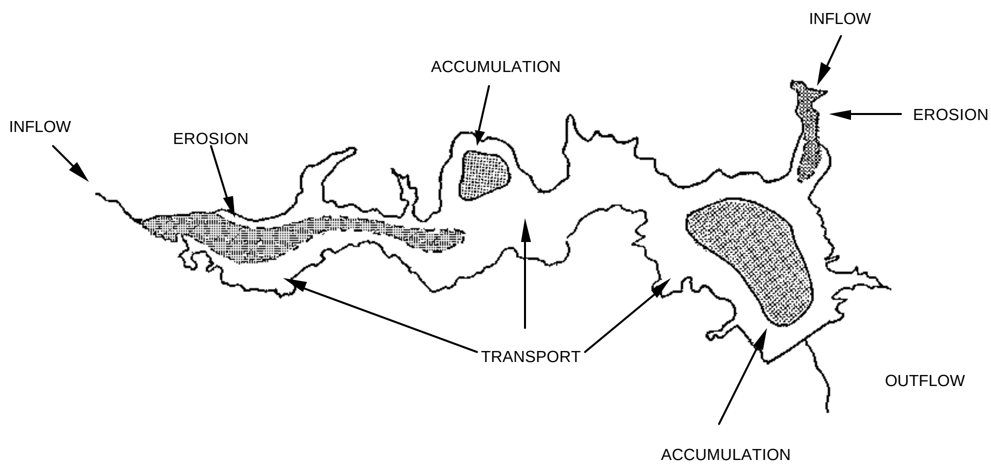


Figure 2.1.3 Zonation of reservoir sediments

2.1.5 DATA ANALYSIS

Data analysis should consist of a well-defined program of data management, verification, preliminary assessments, and appropriate analyses. Data management includes the collection, storage, and distribution of the data and often requires (certainly is enhanced by) the oversight of a single individual to insure quality handling. Typically, data are initially recorded on field sheets or electronically in database managers or data loggers. These data are then transferred to a more permanent database. Verification at this point is best accomplished with visual review of the data in the database and the original data record. Other useful techniques are screening programs that use minimum and maximum values to identify outliers (usually errors during recording or faulty conversions to different units of measurement). For many constituents, environmentally unrealistic values may be used to identify inappropriate values. For instance, a temperature of 80^{o} F would easily be identified as an inappropriate value if the temperature in the database was recorded in ^{o}C and data entry failed to convert the field measurement. Storage of the data is also important especially for distribution to users. A format and media must be selected that is easily transferred. Proper documentation (e.g., units of measure, site location, analytical methods, etc.) should be provided with the data or as a separate file. Preliminary assessments include visual data displays such as scatter plots and simple statistical analyses such as tests for normality, homogeneity, and independence. Once distributions and other assumptions have been evaluated, other statistical techniques (nonparametric and parametric regression analyses, principal components analysis, multivariate analysis, etc.) can be used for more rigorous analysis and hypothesis testing.

An important point about data analysis is that final reporting of the analysis should consider presenting the results clearly and in a format easily interpreted by scientists, managers, and others, depending on the potential audience. Elaborate graphics and sophisticated statistical analyses may be included as an appendix or presented for the more technical reader in a different format.

2.1.6 TOOLS

SAMPLE DESIGN

There are four basic questions in sample design to be considered.

1. What should be sampled? What are the constituents of concern?
2. How many samples should be collected? What is the sample size?
3. When do I collect the samples? What is the sampling frequency?
4. Where do I collect the samples? What sites and depths should be used?

1. Constituent(s)

What should be sampled is a function of the perceived problem. In some case the problem is easily identified and constituents to be monitored can be determined. For example, if hydrogen sulfide odors are a problem, then monitoring for hydrogen sulfide is easily identified (this particular constituent is not easily monitored). However, many processes may contribute to the cause of the problem and a basic knowledge of water quality assists in determining which constituents should be monitored. In the above example with hydrogen sulfide, the origin of the problem is the generation of dissolved sulfide during anoxia in the upstream impoundment. Dissolved oxygen monitoring in the forebay of the impoundment is a logical approach to identifying the development of the problem. Measuring the oxidation-reduction potential may also prove useful in predicting the temporal extent of the problem. More complex problems, such as a fish kill, may require a more detailed assessment. For example several factors may contribute to the sudden death of fish in a reservoir tailwater (e.g., low dissolved oxygen, thermal change, toxic compounds). Problems associated with increased concentrations of reduced metals (e.g., staining, taste and odor), dissolved gasses, and other physico-chemical processes may be repetitive and therefore can be predicted with a well-designed monitoring program. The outcome of identifying the problem (and its source) should provide sufficient information for the design of an enhancement technique.

2. Sample size

Determination of sample size should consider the amount of information needed to address the objectives of the sampling effort with limitations of financial resources often the determining factor. If insufficient funds are available, the option to not conduct the study (not desirable) may have to be used. However, careful consideration of sample size versus data needs may allow an adequate sampling effort. An equation for determining sample size for a simple random sample can be defined by:

$$n = \frac{t^2 s^2}{(r\Theta)^2}$$

where n= sample number, t^2 (ranges between 0.675 and 2.57 for error probabilities between 0.05 and 0.50 with 5 degrees of freedom and restricts the effects of the error probability on sample size, Gaugush 1987), s^2 = variance, r = desired precision, Θ = mean value of the constituent.

Examples of determining sample size using simple random sampling, stratified random sampling, and systematic sampling techniques are presented in Appendix 2.1A.

3. Sample Frequency

The frequency at which samples should be collected is a function of the temporal distribution of the constituent dynamics and is related to the timing and duration of the factors contributing to the water quality problem. A hypothetical time scale suggests relevant response periods of selected constituents during an annual period (Figure 2.1.4). The age of most reservoirs is between 25 and 50 years so trends over this period of time should also be considered. Daily cycles in release and biological activity would require more frequent sampling than events that occur on a seasonal scale (e.g., anoxia, increased concentrations in metals and nutrients). Even more frequent sampling for a shorter duration may be required if microbial processes are a concern. Sample frequency may also be a function of discharge if mass balance evaluations (e.g., loading) are to be conducted and critical periods during low flow is a concern.

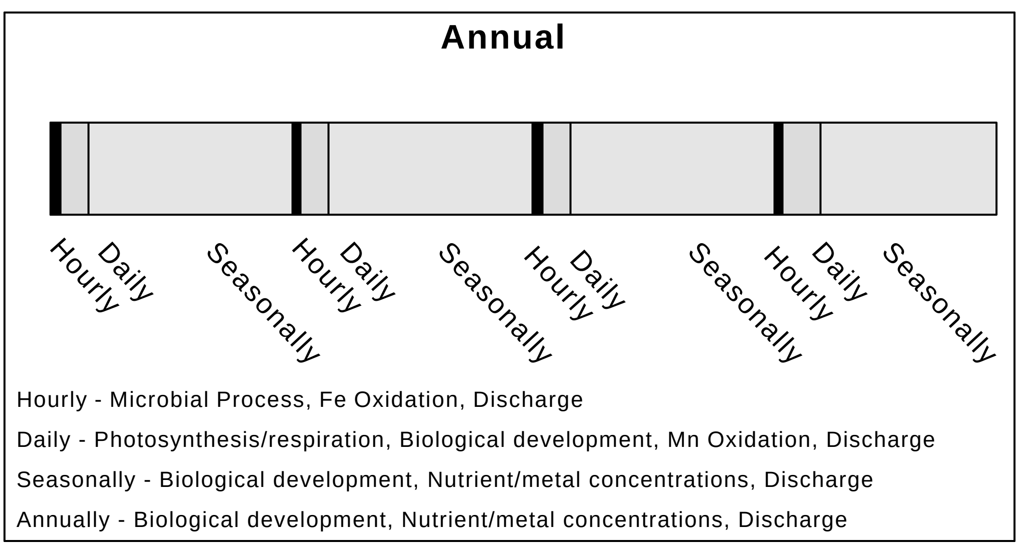


Figure 2.1.4 Hypothetical time scale for selected physical, chemical and biological processes

Using the previous example of hydrogen sulfide, production and subsequent degassing in the discharge, this problem may be considered as a seasonal problem and sampling would be concentrated prior to and during the anoxic period. Using techniques for determining sample size then allows for the determination of a reasonable distribution of sampling effort with the available number of samples. In the case of elevated metals concentrations, this is also a seasonal event but is also related to reservoir operations and sampling effort may focus on critical periods (e.g., low flow or maximum concentrations) or on total loads (more frequent measurements during increased discharge). For episodic events, i.e., a fish kill, intensive sampling may be the best approach and frequency is again a function of sample size (often limited by resources).

Other considerations for sampling frequency include the response of the water resource to changes in pollutant source, the magnitude of the minimum amount of change required for detection (MDC), the system variability, and the probability of detecting a trend (statistical power) (USEPA 1991). MDC is defined as the minimum change in a water quality parameter over time that is considered statistically significant. The MDC can be estimated from historical records to aid in determining the required sampling frequency and to evaluate monitoring feasibility (Spooner et al. 1987). An example of sample allocation is provided later in this chapter in the tools section.

4. Site Location

Where to sample is also a function of the perceived problem and sample frequency (or monitoring program) designed to identify the source of the problem and may be a single site, established fixed locations, or several stations determined in the field as in an intensive study.
For fixed locations, representativeness (discussed in more detail later in this chapter) and accessibility are important considerations. The sites selected must represent conditions for the lateral transect of the section or include variability estimates and must be accessible for sample and data collection and instrument calibration and maintenance. The use of established USGS gaging stations is a good choice for correlation to discharge and possible access to a secured site. While bridges, boat ramps, and recreation sites are often easily accessed, representativeness at these sites should be evaluated, particularly if the flow is highly variable (due to islands, bridge supports, abutments, etc.) or channel morphometry has been modified (e.g., protected or dredged areas for boat launching or recreation). Other considerations include changes in local geology which may effect certain chemical constituents, changes in land use, differences in riparian conditions, and locations of groundwater, tributary, and point sources. All of these factors could contribute to changes in longitudinal distributions and can be addressed with appropriate site location (e.g., upstream and downstream comparisons between sites). A final note on site location is to mention that sampling from a habitat perspective may be a useful approach. For instance, if the fisheries or benthic community is the focus of the study, then sampling in a variety of habitats at several locations may be a better approach than more stations distributed longitudinally or more frequent sampling at selected sites. Site location is a characterization process determined by the objectives of the study.

Gaugush (1993) developed sampling design software specifically for water quality sampling in reservoirs and tailwaters and the software and user's manual are included with this manual. One of the applications of this software includes a trade off analysis based on sample size (or allocation of constituents) versus available budget and the change in certainty associated with redistribution or reduction of sample allocation and size. This software and an example application are provided in Appendix 2.1B.

2.1.7 WATER QUALITY MODELS

Numerous water quality models exist for assessing rivers and reservoirs. These models provide 1, 2, and 3 dimensional descriptions of water quality processes including hydrodynamics, thermal attenuation, chemical cycling, and biological interactions and dynamics. A partial list of models and a brief description is provided in Table 2.1.1.

2.1.8 FATE AND TRANSPORT MODELING FOR CONTAMINANTS

The movement of contaminants in an aquatic system may occur through biological, chemical, or physical processes. Models such as WASP4 (Ambrose et al. 1988) and CE models such as RECOVERY and ICM/TOXI (described below) may be used to simulate transport and fate of toxic pollutants in aquatic systems such as lakes, reservoirs, and estuaries.

The RECOVERY model is time-varying with a well-mixed, zero-dimensional water column underlain by a vertically stratified, one-dimensional (1D) sediment column (Boyer et al. 1994). RECOVERY contaminant fate processes include: water column and sediment sorption and decay; water column volatilization; sediment burial and resuspension; water column settling; sediment and pore water advection; and pore water diffusion among sediment layers and across the sediment-water interface. RECOVERY assumes a steady-state solids balance for one class of solids where the user specifies the suspended solids concentration and two of the three rates for burial, settling, and resuspension, and the model computes the third rate. Sediment porosity can vary over depth but is constant over time. RECOVERY uses an Eulerian framework with a fixed number of layers. Thus, burial and resuspension result in vertical advection of sediment-bound and pore water contaminant relative to the fixed surficial sediment reference. The 1D, total contaminant concentration equation for the sediment bed is solved with a Crank-Nicholson finite difference representation.

ICM/TOXI was developed from the 3D eutrophication model, CE-QUAL-ICM (Cerco and Cole 1995), which was originally developed during a study of Chesapeake Bay (Cerco and Cole 1993) and was modified (Wang et al. 1997) for application to trace contaminants. The contaminant version is referred to as CE-QUAL-ICM/TOXI, or simply ICM/TOXI. The chemical kinetic algorithms included in ICM/TOXI were based on those of the Water Quality Analysis Program, WASP, (Ambrose, Wool and Martin 1993).

Table 2.1.1 Selected Water Quality Models		
Waterbody	Model	Description
River	STEADY	1-D longitudinal steady-state, temperature, DO, and BOD
	QUAL2E	1-D longitudinal steady flow stream water quality model (EPA).
	FLUX	1-D loading model
	CE-QUAL-RIV1	1-D dynamic flow, time-varying stream hydraulic (RIV1H) and water quality (RIV1Q) models.
	TWQM	Tailwater Quality Model, 1-D steady flow, steady-state stream water quality model for reservoirs and tailwaters. Includes reduced substances (e.g. iron, manganese, and sulfides).
Reservoir	PROFILE	1-D vertical 2 layer (mixed upper layer and hypolimnion) steady-state model, calculates oxygen deficit rates.
	BATHTUB	1-D longitudinal 2 layer (mixed upper layer and hypolimnion) steady-state empirical eutrophication model.
	CE-THERM-R1	1-D vertical reservoir model for temperature.
	CE-QUAL-R1	1-D vertical reservoir model for water quality.
	WESTEX	1-D vertical reservoir model for temperature and conservative constituents.
	CE-QUAL-W2	2-D laterally-averaged, longitudinal, vertical, hydrodynamic and water quality model for reservoirs, estuaries, and other 2-D water bodies.
	SELECT	1-D vertical, steady-state model for predicting the selective withdrawal outflow distribution and release concentrations for reservoir outlets.
Others	HEC-5Q	1-D vertical (reservoir) and 1-D longitudinal (stream) reservoir systems operation model that includes water quality.
	CE-QUAL-ICM	1-D, 2-D, and/or 3-D water quality model. Must be linked to hydrodynamic model output.
	RECOVERY	Zero-Dimensional (water column) PC based screening model to assess the impact of contaminated bottom sediments on surface waters. The bottom sediments are modeled with 1-D vertical layers.

Physical processes such as erosion, transport, and deposition, which may be more important in riverine systems, may be described with sediment transport models ranging in complexity from HEC-6 (USACE 1990) and SEDZL frameworks (Ziegler and Lick 1986) to screening level approaches such as the in-place pollutant export model (IPX) developed by Velleux et al. 1994.

2.1.9 REFERENCES

Alexander, C. M. 1995. "A summary of the distribution and effects of polychlorinated biphenyl (PCB) contamination in aquatic biota in Twelve Mile Creek and Lake Hartwell, South Carolina," in Water Quality '94, Proceedings of the 10th Seminar, 15-18 February 1994, Savannah, GA, Miscellaneous Paper W-95-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Ambrose, R.B., Wool, T.A., Connolly, J.P., and Schanz, R.W. 1988. "WASP4, a hydrodynamic and water quality model - model theory, user's manual, and programmer's guide." EPA/600/3-87/039, U.S. Environ. Protection Agency, Athens, GA.

Ambrose, R. B., Wool, T. A., and Martin, J. L. (1993). "The Water Quality Analysis Simulation Program, WASP5; Part A: Model documentation," U.S. Environmental Protection Agency, Center for Exposure Assessment Modeling, Athens, GA.

Boyer, J. M., Chapra, S. C., Ruiz, C. E., and Dortch, M. S. (1994). "RECOVERY, a mathematical model to predict the temporal response of surface water to contaminated sediments," Technical Report W-94-4, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Cairns, J, Jr., McCormick, P. V., and Niederlehner, B. R. 1993. "A proposed framework for developing indicators of ecosystem health," *Hydrobiologia*, 263:1-44.

Carlson, R. E. 1977. "A trophic state index for lakes," *Limnol. Oceanogr.,* 22:361-369.

Cerco, C. F., and Cole, T. (1993). "Three-dimensional eutrophication model of Chesapeake Bay, *J. Environ. Eng.*, Am. Soc. Civil Eng., 119(6), 1006-1025.

Cerco, C. F., and Cole, T. M. (1995). "User's guide to the CE-QUAL-ICM three-dimensional eutrophication model", Technical Report EL-95-15, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Cremeans, W.L., Meyer, R.M., and Kincaid, G.P. 1995. "High-speed system for synoptic assessment of riverine near-surface water quality conditions and spill response," in Water Quality '94, Proceedings of the 10th Seminar, 15-18 February 1994, Savannah, GA, Miscellaneous Paper W-95-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Gaugush, R. F. 1987. "Sampling Design for Reservoir Water Quality Investigations," Instruction Report E-87-1, U. S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Gaugush, R. F. 1993. "Sampling Design Software User's Manual," Instruction Report W-93-1, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Heiskary, S. A. and Walker, W. W., Jr. 1988. "Developing phosphorus criteria for Minnesota Lakes," *Lake and Reservoir Management*, 4(1):1-9.

Hunsaker, D. T. and Carpenter, D. E. (Eds). 1990. "Environmental monitoring and assessment program: Ecological indicators." Office of Research and Development, United States Protection Agency, Research Triangle Park, NC.

Intergovernmental Task Force on Monitoring Water Quality. 1995. "The strategy for improving water-quality monitoring in the United States," Final Report, Appendix E. U.S. Geological Survey, Reston, VA.

Karr, J. R., Fausch, K .D., Yant, P. R., and Schlosser, I. J. 1986. "Assessing biological integrity in running waters: a method and its rationale," Illinois Natural History Survey, Special Publication 5, 28 p.

OECD (Organization for Economic Cooperation and Development). 1982. "Eutrophication of waters. Monitoring, assessment and control." Final report. OECD Cooperative Programme on Monitoring of Inland Waters (Eutrophication Control), Environment Directorate, OECD, Paris, 154 p.

Osmond, D. L., R. W. Gannon, J. A. Gale, D. E. Line, C. B. Knott, K. A. Phillips, M. H. Turner, M. A. foster, and D. E. Lenning. 1997. "WATERSHEDS: A Decision Support System for Watershed-Scale Nonpoint Source Water Quality Problems. *Journal of the American Water Resources Association*, 33, 327.

Plafkin, J. L., M. T. Barbour, K. D. Porter, S. K. Gross, and R. H. Hughes. 1989. "Rapid Bioassessment Protocols for Use in Streams and Rivers: Benthic Invertebrates and Fish," US Environmental Protection Agency, Office of Water, Washington, DC. EPA/444/4-89-001.

Renner, R. 1997. "Rethinking water quality standards for metals toxicity," *Environ. Sci. and Tech.*, 31(10):466A-468A.

Ryder, R. A. 1965. "A method for estimating the potential fish production of north-temperate lakes," Trans. Am. Fish. Soc., 94:214-218.

Shannon, C. E. and Weaver, W. 1949. *The mathematical theory of communication*, University of Illinois Press, Urbana.

Spooner, J., Jamieson, C. J., Maas, R. P. and Smolen, M. D. 1987. "Determining Statistically Significant Changes in Water Pollutant Concentrations," *Journal of Lake and Reservoir Management*, 3:195-201.

Terrell, C. R. and Perfetti, P. B. 1989. "Water quality indicators guide: surface waters," U.S. department of Agriculture, Soil Conservation Service, SCS-TP-161, Washington D.C.

U. S. Army Corps of Engineers. 1990. "Status and new capabilities of computer program HEC-6: scour and deposition in rivers and reservoirs." Tech. Paper No. 129, U.S. Army Corps of Engineers, Hydro. Engrg. Ctr., Davis, CA.

U. S. Environmental Protection Agency. 1990. Biological Criteria: National Program Guidance for Surface Waters. EPA-440/5-90-004. U.S. Environmental Protection Agency, Washington, DC.

U. S. Environmental Protection Agency. 1991. Biological Criteria: Research and Regulation. Proceedings of a Symposium. EPA-440/5-91-005. Environmental Protection Agency, Office of Water, Health and Ecological Criteria Division, Washington, DC.

U. S. Environmental Protection Agency. 1991. *Watershed Monitoring Report for Section 319 National Monitoring Program Projects.* U.S. Environmental Protection Agency, Office of Water, Washington, D.C.

U. S. Environmental Protection Agency. 1992. Procedures for Initiating Narrative Biological Criteria. EPA-882-B-92-002. U.S. Environmental Protection Agency, Office of Science and Technology, Washington, DC.

U. S. Environmental Protection Agency. 1996a. Biological Assessment Methods, Biocriteria, and Biological Indicators: Bibliography of Selected Technical, Policy, and Regulatory Literature. EPA-230-B-96-001. Office of Policy, Planning, and Evaluation, U.S. Environmental Protection Agency, Washington, DC.

U. S. Environmental Protection Agency. 1996b. Biological Criteria: Technical Guidance for Streams and Small Rivers. EPA-882-B-96-001. U.S. Environmental Protection Agency, Office of Science and Technology, Washington, DC.

U. S. Environmental Protection Agency. 1996c. Summary of State Biological Assessment Programs for Streams and Rivers. EPA-230-R-96-007. Office of Policy, Planning and Evaluation, U.S. Environmental Protection Agency, Washington, DC.

U. S. Environmental Protection Agency. 1998. Lake and Reservoir Bioassessment and Biocriteria, Technical Guidance Document. EPA-841-B-98-007. Office of Water, U.S. Environmental Protection Agency, Washington, DC.
U.S. Geological Survey. 1998. "Water quality in the Apalachicola-Chattahoochee-Flint River Basin, Georgia, Alabama, Florida, 1992-1995," USGS Circular 1164.

Velleux, M., Gailani, J., and Endicott, D. 1994. "User's guide to in-place pollutant export (IPX) water quality modeling framework." Draft Rep., U.S. Environmental Protection Agency, Large Lake Res. Station, Grosse Isle, MI.

Wang, P. F., Martin, J. L., Wool, T., and Dortch, M. S. (1997). "CE-QUAL-ICM/TOXI, a three-dimensional contaminant model for surface water: Model theory and user guide," draft Instruction Report, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Wilhm, J. L. 1967. "Comparison of some diversity indices applied to populations of benthic microorganisms in a stream receiving organic wastes," *J. Water Poll. Cont. Fed*., 39:1673-1683.

Yagecic, J. and Moy, P.B. 1995. "Effects of motorized watercraft on sediment quality at Fox Chain O'Lakes in northern Illinois," in Water Quality '94, Proceedings of the 10th Seminar, 15-18 February 1994, Savannah, GA, Miscellaneous Paper W-95-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Ziegler, C.K. and Lick, W. 1986. "A numerical model of the resuspension, deposition, and transport of fine-grained sediments in shallow water." Rep. ME-86-3, Dept. Of Mech. Engrg., Univ. of California-Santa Barbara, Santa Barbara, CA.

2.2 ASSESSMENT OF WATERSHED PROCESSES

2.2.1 THE NONPOINT SOURCE MANAGER'S GUIDE TO WATER QUALITY AND LAND TREATMENT MONITORING

Problem definitions are based on the designated use of the water resource. This section will discuss designated uses and how pollutants and conditions can threaten a use or impair a use. Please refer to the report "The Nonpoint Source Manager's Guide to Water Quality and Lane Treatment Monitoring" at the end of this chapter.

2.2.2 DATA ANALYSIS

2.2.2.1 Linking Water Quality Trends with Land Treatment Trends: The Rural Clean Water Program Experience

Land use and land management affect the type and amount of nonpoint source (NPS) pollution entering water bodies. Improvements in land management (also referred to as land treatment) are necessary to reduce the delivery of pollutants to impaired or threatened water resources. Documentation of the magnitude of water quality improvements from changes in land management, for at least a few projects in each part of the country, is essential to provide feedback to project coordinators and state, regional, and national policy makers. Such feedback enhances the development and implementation of land treatment programs that effectively reduce delivery of pollutants causing water quality impairment. In addition, demonstration that land treatment is effective in reducing NPS pollution and improving water quality tends to increase political and economic support for NPS pollution control measures.

Historically, it has been difficult to demonstrate the relationship between land treatment and water quality changes, at least in part because of a lack of well-designed water quality and land treatment monitoring efforts. Two goals must guide the design of monitoring networks and data analysis in programs and projects designed to link water quality changes with implementation of best management practices (BMPs): 1) detection of significant (or real) trends in both water quality and land treatment and 2) linking or associating water quality trends with land treatment trends.

This fact sheet outlines the principles for development of effective monitoring designs, and describes the land treatment and water quality monitoring elements necessary for linking land treatment or land use modifications with water quality changes. These monitoring elements are essential for successful experimental watershed projects designed to document the relationship between land treatment and water quality changes.

Many of the recommendations for monitoring discussed in this fact sheet are based on the 15-year Rural Clean Water Program (RCWP), an experimental, agricultural watershed, NPS pollution control program that combined land treatment and water quality monitoring in a continuous feedback loop to document NPS control effectiveness (Gale et al., 1993; Spooner and Line, 1993).

2.2.2.2 Documenting a Cause-and-Effect Relationship

Documenting that water quality changes at a watershed scale were caused by implementation of BMPs is difficult. Not only must a strong correlation be established, but the observed changes must be repeatable over time and space in an experimental manner. The only major changes made in the watershed during the evaluation period should be changes in land treatment. Observed changes in water quality should match predicted pollutant reductions based on the estimated land treatment effectiveness. Some projects have been able to document a strong relationship, increasing confidence that appropriate land treatment can result in (cause) improved water quality. The stronger the relationship, the more likely it is that a cause-and-effect relationship exists and that water quality changes are caused by changes in land treatment rather than other factors.

An association (statistically significant correlation or relationship) between land treatment and water quality changes is required to demonstrate a cause-and-effect relationship. As the implementation of land treatment (specifically BMPs) occurs, improvements in water quality are observed. However, an association by itself is not sufficient to infer a cause-and-effect relationship. Other factors not related to BMP implementation may be causing the changes in water quality, such as changes in land use or rainfall. If, however, the association is consistent and responsive and has a mechanistic basis, causality may be supported (Mosteller and Tukey, 1977).

Consistency means that the relationship between the measured variables (such as total phosphorus and acres treated with the nutrient management BMP) holds in each data set in terms of direction and degree. A consistent, multi-year, improving trend in water quality after BMP implementation provides evidence needed to attribute water quality improvements to land treatment. Improvements in multiple watersheds treated with systems of BMPs provide strong evidence that water quality improvements resulted from land treatment.

Responsiveness signifies that as one variable changes in a known, experimental manner, the other variable changes similarly. For example, as the amount of land treatment increases, further reduction of pollutant delivery to the water resource is documented.

Mechanistic means that the observed water quality change is that which is expected based on the physical processes involved in the installed BMPs. For example, based on knowledge of absorption and solubility of nutrients, greater reduction of nutrient delivery to the water resource might be predicted as the result of implementation of the manure management BMP than a soil erosion control practice alone.

2.2.2.3 Elements of Monitoring Needed to Link Land Management Modifications with Water Quality Changes

Experimental Designs for Water Quality and Land Treatment. An appropriate experimental design for water quality and land treatment monitoring is essential to document a clear relationship between land treatment and water quality changes. The best designs to demonstrate linkage are those that can isolate the effects of the land treatment from other land use and climatic changes. Such designs include: 1) paired watershed (Clausen and Spooner, 1993); 2) upstream-downstream sites monitored before, during, and after land treatment; and 3) multiple watershed monitoring.

The paired watershed design is the best method for documenting BMP effectiveness in a limited number of years (three to five). Two or more similar subwatersheds (drainage areas) are monitored before and after implementation of BMPs in one of the subwatersheds (the treatment subwatershed). Paired drainage areas should have similar precipitation and runoff patterns and should exhibit a consistent relationship in terms of the magnitude of pollutant losses with changes in hydrology and climate. Analysis of paired pollutant data from treatment vs. control areas should show a statistically significant correlation. Ideally, a paired watershed monitoring program is characterized by:

- Simultaneous monitoring at the outlet of each drainage area;
- Monitoring prior to land treatment to record the relative hydrologic response of each drainage area (calibration period);
- Calibration period of one to three years, depending on the consistency of the data relationships between drainage areas;
- Subsequent monitoring where at least one drainage area continues to serve as a control (that is, receives significantly less land treatment than the other drainage area); and
- Similar land management in both drainage areas both before and after BMP implementation (for example, similar crops), except for BMPs implemented in the treatment drainage area.

Land Management and Water Quality Monitoring Before and After BMP Implementation. Monitoring for several years both before and after BMP implementation is essential for documentation of water quality changes. The pre-BMP period is the time prior to installation of new land treatment practices. Monitoring of water quality and land use prior to BMP implementation is required to establish baseline data for statistical comparison with post-implementation data. The post-BMP period starts once BMPs have been implemented on critical areas and are reducing pollutant delivery to the water resource.

Sampling frequency and collection must be consistent across seasons and years. Year-to-year variability is often so large that at least two to three years each of pre- and post-implementation monitoring is required to indicate a consistent water quality change following implementation and maintenance of BMPs. Documentation of changes over multiple years increases confidence that observed water quality improvements are due to land treatment.

Short-term monitoring is seldom effective because climatic and hydrologic variability can mask water quality changes. However, in small watersheds affected by relatively few large pollutant sources, the monitoring period may be shorter. Longer duration monitoring is necessary where water quality changes are likely to occur gradually, such as large watersheds with lakes in which lag times may occur due to buffering effects of long hydraulic residence times and pollutant recycling.

Quantitative Monitoring of Land Management. The importance of recording the amount and type of land treatment cannot be overlooked when trying to establish documented water quality improvements. Best management practices must be targeted to treat specific sources of pollutants causing the water quality impairment; these pollutants, in turn, must be monitored in the water resource. A high level of appropriate NPS pollution control implementation in critical areas is usually required to achieve substantial water quality improvements.

Monitoring of land treatment and land use is needed to quantify the pollutant reduction impacts of BMPs. Quantitative monitoring of BMP implementation facilitates documentation of land treatment trends and is a necessary step in linking water quality to land treatment. Methods of reporting and quantifying land treatment and land use should be consistent throughout a project.

Careful planning is required to determine which land treatment variables should be monitored to best reflect the extent of actual changes in agricultural practices. Land treatment data must be reported in quantitative units that reflect BMP effectiveness and changes from previous practices. Examples of quantitative units include: application method, tons of manure spread per acre, pounds of fertilizer applied per acre, acres served by each BMP, and acres served by each BMP system. The acres served unit includes all treated acres (those acres with actual implementation) plus all acres whose pollutant delivery is being reduced by the BMP. Documenting the assumptions used in calculating the acres served is important so that these units can be calculated consistently from year to year, thus ensuring valid year-to-year comparisons.

When reporting acres served, care should be taken to avoid double counting acres when multiple BMPs are serving the same acres, as this could artificially inflate the reported number of acres served. In addition, correction should be given for differences in the effectiveness of the BMPs in controlling pollutant delivery.

Operation, management, and maintenance of BMPs should be tracked because these factors affect BMP effectiveness and, therefore, the water quality impacts of the land treatment.

Changes in land use should be recorded to help isolate the water quality changes associated with the NPS controls from water quality changes due to other land use factors. Land use modifications that affect water quality include acres converted from row crops to pasture (permanently or based on rotation), set-aside acres, changes in the number of animals or animal units per acre, closure of animal operations, changes in impervious land areas, implementation of soil and water conservation practices not being recorded as part of the project, and changes in non-agricultural land uses.

Matching of Land Treatment and Water Quality Data on a Spatial (Drainage) Scale. Land treatment data must be collected on a hydrologic or drainage basis so that the land area being tracked corresponds to the drainage area served by each water quality monitoring station. Water quality and land treatment data must be matched if water quality changes are to be attributed to BMP implementation.

Linkage of land treatment and water quality impacts can be made at different spatial scales (such as farm field, subwatershed, or watershed). Spatial scale should be determined based on project goals and desired interpretations. In general, the larger the drainage area, the harder it is to identify and quantify a water quality - land treatment linkage. Water quality changes are more likely to be observed at the subwatershed than watershed level. Confounding effects of external factors, other pollutant sources, and scattered BMP implementation are minimized at the subwatershed level. If the goal is to document changes at the entire watershed level, a monitoring station must be located at the watershed outlet.

Matching of Land Treatment and Water Quality Data on a Temporal Scale. Water quality and land treatment data should be collected during the same time periods so both data sets are temporally related. Actual implementation of land treatment needs to be recorded at least seasonally or annually. Land treatment data (such as timing of manure or commercial fertilizer applications, construction of a lagoon storage structure, or a dairy closure) should be collected more frequently than annually or seasonally if the effect on water quality is more short-term or has a large, immediate impact.

Water quality samples are usually collected weekly or biweekly. These data do not have to be summarized on the same time scale as the land treatment data; land treatment data can be added to the trend analysis as repeating explanatory variables. Alternatively, water quality data can be aggregated to the same time scale as the land treatment data for analysis. Data aggregation is particularly useful for plotting and explanatory data analysis.

Matching Monitored Pollutants with Pollutants Addressed by Land Treatment. Pollutants monitored at water quality stations must correspond to pollutant(s) being treated by the BMP systems implemented.

Monitoring Explanatory Variables. Accounting for all major sources of variability in water quality and land treatment data increases the likelihood of isolating water quality trends resulting from

BMPs. Correlation of water quality and land treatment changes by itself is not sufficient to infer causal relationships. Other factors not related to BMPs may be causing water quality changes, such as changes in animal numbers, cropping patterns, land uses, known pollutant sources, or amount of impervious land surface; season; stream discharge; precipitation; ground water table depth; salinity; or other climatic or hydrologic variables. Factoring explanatory variables into trend analyses yields water quality trends closer to those that would have been measured had no changes in climatic or other explanatory variables occurred over time. Accounting for variability in water quality due to known causes also decreases variation in adjusted water quality data, facilitating documentation of statistically significant trends. Explanatory variables should be monitored at the same frequency as the principle water quality variables.

2.2.2.4 Summary

A good experimental design for water quality and land treatment monitoring is essential in order to provide clear documentation of the relationship between land treatment and water quality changes. The paired watershed monitoring design can best demonstrate the relationship between land treatment and water quality in the shortest period of time.

To determine if the trends in water quality match the mechanistic prediction of trends, pre- and post-BMP implementation monitoring and data analysis must combine water quality, land treatment, and land use data on suitable spatial and temporal scales. Incorporation of explanatory variables facilitates isolation of water quality changes that result from land treatment.

2.2.3 CASE STUDY

RCWP examples are available in "Evaluation of the Experimental Rural Clean Water Program," shown at the end of this chapter.

2.2.4 REFERENCES

Clausen, J.C. and J. Spooner. 1993. *Paired Watershed Study Design*. Office of Water, US Environmental Protection Agency, Washington, DC. EPA 841-F-93-009. 8 p.

Coffey, S. W., Spooner, J., and Smole, M. D. 1995 (Feb). "The Nonpoint source manager's Guide to Water Quality and Lane Treatment Monitoring," North Carolina Cooperative Extension Service, North Carolina State University, Raleigh, NC, under a grant from the Assessment and Watershed Protection Division, US Environmental Protection Agency, Washington, DC.

Gale, J. A., Line, D. E., Osmond, D. L., Coffey, S. W., Spooner, J., Arnold, J. A., Hoban, T. J., and Wimberley, R. C. 1993. "Evaluation of the Experimental Rural Clean Water Program," North

Carolina Cooperative Extension Service, North Carolina State University, Raleigh, NC, under a grant from the Nonpoint Source Control Branch, US Environmental Protection Agency, Washington, DC.

Mosteller, F. and J.W. Tukey. 1977. *Data Analysis and Regression: Second Course in Statistics*. Addison-Wesley Pub. Co., Reading, MA 588 p.

Spooner, J., Line, D. E., Coffey, S. W., Osmond, D. L., and Gale, J. A. 1995 (Mar). "Linking Water Quality Trends with Land Treatment Trends; The Rural Clean Water Program Experience," North Carolina Cooperative Extension Service, North Carolina State University.

2.3 ASSESSMENT OF RESERVOIR PROCESSES

INTRODUCTION

This section includes a restricted range of assessments of lakes. Risk assessment, for example, is not addressed. Instead, this section addresses more general questions of assessment of problems, assessment of results, and assessment of solutions.

In practice, there are simple but important statements of the problems for lake assessment. What tools should be used? What methods should be used? The tools can be hardware or computer programs and our choices are well-known and familiar. The methods, however, are varied and greatly dependent on the particular application. Methods employ the tools to interpret or investigate ideas or observations and methods are therefore also dependent on the underlying theory, if any, for the investigation.

Where the problems are simple, **?**do nutrient concentrations exceed regulatory standards?" the process may also be simple. But even for simple questions, the answer in complex aquatic systems may involve attention to many details and many technical decisions.

The sources of complexity in lake assessment usually come from two categories of variation of lake characteristics. Temporal variation is the first and this requires repetitive observation for assessment. Spatial variation is the other category and this requires multiple observations in space. They are not incompatible and can combine to require elaborate, expensive assessment efforts.

Temporal complexity is obvious. There are common, universally-recognized periodic processes in aquatic systems. Primary among these is the diel (24 hour) period associated with the earth's rotation. This is superimposed on the annual or seasonal cycle. Biota can have even more novel cycles with larval forms growing continuously for months followed by sudden, brief, synchronous emergences in adult form for mating and egg laying. Obviously the way we choose to observe such different events must be different and designed to optimize the clarity of our observations while minimizing our effort. This is true for all environments, not just for lakes and reservoirs.

Spatial complexity occurs in three dimensions, all of which are commonly sampled in reservoirs. Reservoirs often have very complex shapes. As a result of this factor, there may be wide ranges of conditions in a reservoir and our ability to describe or assess them depends on our skill of observation. Do we emphasize tributary embayments or deeper waters where chemical processes may be more complex? How do we fairly represent all of the various locations? Again, the way we choose to observe must optimize the clarity of our observations while minimizing the depletion of our resources.

The answer to these inevitable questions which determine our assessment success are both theoretical and practical. In this chapter are presented both views with emphasis on the practical view.

2.3.1 LAKE ASSESSMENT, THE FIRST STEP

To complete the task of lake assessment, there are implicit components that must be addressed. Often these are violated. For example, an assessment might not have the luxury of a data collection effort because abundant data are already available. Or else there are insufficient resources to acquire more data. (In this example, one can substitute ‘observation’ for ‘data’) In this case all of the practical questions leading to the formation of a database are skipped and the last step - data analysis - is all that is left for completion. If there are abundant appropriate data this is the ideal situation for lake assessment. If there are not abundant data, this is called ‘reality’.

The process of collecting needed observations is important and needs to planned as carefully as possible. There are two extremes for this situation, a thorough statistically-sound design versus a simple minimal design specifically for the question or problem at hand.

2.3.2 SAMPLING DESIGN

As with tailwater assessment the first step in this process is clearly identifying the problem or question. This will allow definition of the objectives. The objectives are an important reference during the entire process of sampling design. Rational consideration of the sampling requirements necessary to accomplish the objectives will facilitate early identification of needed resources, or if the objectives are even possible.. or not.

The design procedure involves answering several questions:

1. What needs to be observed (measured)?
2. How often and where?
3. How many replicates?

An objective way to answer these is to apply statistical methods to pre-existing knowledge of the lake or reservoir. Fortunately there are abundant resources for a statistically-sound sampling design. Recent published works include Montgomery (1997), Green (1979), Gilbert (1987), and Gaugush (1987 and 1993). These are specialized for sampling design and build on knowledge of basic statistical concepts. For those basic concepts other standard references include Neter and Wasserman (1974), Steele and Torrie (1980) or Sokal and Rohlf (1969). Gaugush, in particular, addresses the requirements for reservoir water quality investigations and has published (Gaugush 1993) software to aid in the design. All of these approaches incorporate statistical considerations of intended objectives; trend or prediction?, statistical confidence, QA and QC considerations. Pragmatically, these statistical approaches tend to assume that some minimal standard (for example, a confidence limit) exists that must be exceeded in the results of the assessment. Such limits of effort are nearly always present. In these treatments of statistical sampling design there are dominant issues of meeting statistical assumptions (especially for predictive exercises) and the various means (transformations, etc.) that may be employed later in the analyses.

2.3.2.1 The Practical Reality

In practice, studies and datasets rarely meet all of the demands set by statistical design. This does not mean that the data is without value but that its value depends more on sound analysis. Even where a study has been meticulously designed to meet all needed criteria, circumstances (an extreme storm event, for example) can quickly cause previous assumptions not to be met. (It is surprising how common 100- and 500-year events seem to be). For the purpose of data analysis, a safe assumption is that no particular design has been met.

Assessment methods are also well documented. There are two sources for these and they include theoretical references such as Thomann and Mueller (1987), Stauffer (1981), Reckhow (1979), or numerous EPA references such as The Lake and Reservoir Restoration Guidance Manual (Olem and Flock, eds. 1990). The other source of documentation of assessment methods include the actual lake assessments. These are instructive for their diversity of problems and methods, the creative solutions discovered, and their diversity of detail and success.

The simplest lake assessment situation is a small lake which is not morphometrically complex and which has a central deeper region. For the purposes of assessment of loading, there would only be one well-defined inflow and one well-defined outflow. There are a large number of lakes that approximately fit this description, few of which ever need intensive assessment efforts.

The more complex cases seem to demand the greatest assessment resources. These would be larger, deeper reservoirs with complex morphometries, numerous inflows from varying sources, greatly variable in-lake conditions depending on spatial location, and outflows complicated by seasonally changing inflows, power demands, and fisheries habitat needs. If periodic reversal of outflows was added to this list, several pumped-storage reservoirs would fit the description.

Common and recent lake issues have included; eutrophication, acid rain, fisheries habitat, release quality, municipal water supply (quality and quantity), industrial use, non-point sources of pollutants, and others. If these were applied to the most complex lake examples, the methods of assessment would be very numerous. However, some would be common to all issues because they are either easy to do or they are fundamental to nearly all issues. The limnological basics which lend understanding to physical energy exchanges, water movements, and simple chemical processes related to temperature, dissolved oxygen, and the carbon system relate to most of those fundamental methods.

The practical reality is that we tend to sample as much as is feasible, usually limited by resources. Such sampling is often performed in a non-random manner, often biased by location and timing (we rarely sample lakes at night or during storms). Under these circumstances making the best of the results requires very practical methods of field collection and data analysis, statistical and otherwise.

2.3.2.2 The Methods of Assessment - Field Collections

Here the methods of lake assessment are restricted to those procedures likely to be performed in the field. These include *in situ* measurements as well as collecting water or biological samples for later analysis.

The number of lake characteristics that we can sample using *in situ* instrumentation is growing rapidly. These characteristics include several parameters that are nearly always measured: temperature, dissolved oxygen, pH, specific conductance, depth, location and time, and secchi transparency. Sometimes included are also turbidity, light intensity, oxidation reduction potential, and a growing list of specific ions. The technologies are mature for most of the basic parameters and instrumentation for measuring them has become very refined. Most improvements in recent instrumentation are related to ease of use or digital manipulation of the data. Occasionally, new technologies are added - optical or solid state sensors, for example.

These instruments are great improvements over earlier ones because the assessment process can begin in the field. Quick, accurate assessment of thermal or chemical trends is possible and field decisions can be made or changed to optimize the return of useful data per unit effort. For example, unexpected thermal trends can be more intensively studied at the time that they are observed or else additional chemical observations can be added depending on decisions in the field.

This latitude of judgement contains risks. The first step toward all subsequent analyses is the field collection step and if this is inaccurate or the result of poor judgement then all subsequent steps in analysis or assessment are compromised. Field personnel must understand the measurement technology and the limnological processes relevant to the parameters measured. In addition, the ease of use allowed by modern instrumentation can convey false confidence in measurements - confidence which did not exist for more ‘primitive’ instrumentation. There is still much good work that can be done with basic and technologically simple methods.

The two best references for field and laboratory analyses are Standard Methods (Greenberg 1995) and the EPA manual (EPA 1979). There are other sources of methods and these include Strickland and Parsons (1972), Wetzel and Likens (1991) as well as manuals specific to instrument manufacturers such as Hydrolab, Bran Luebbe, YSI/Endeco, Perkin Elmer, and others.

2.3.3 SAMPLING CONSIDERATIONS

Many issues to be considered prior to sampling have already been stated or discussed. After the problem or question has been identified, the design of a sampling program to address the problem will require knowledge of the range of lake depths, the number of significant inflows, the presence of large or unique tributary embayments, and (if possible) pre-existing knowledge of spatial trends in the lake.

In the absence of statistical design, or where specific needs are restricted to certain areas of a lake, the number of depths of observations is important. Too many depths will require large amounts of sampling time and too few depths may not resolve the depth structure sufficiently to address the question. This becomes more true for deeper lakes in which significant investments of time are made for relatively few depth ‹profiles'. If pre-existing knowledge allows, it is possible to vary the sample depth increment to economize on effort and time. For example, increments of one meter in depth in the region of the thermocline can yield to five- or ten-meter increments in deeper waters if the temperature does not significantly change. However, this may lead to problems of data management or data analysis if the variety of sampling is great.

In small, morphometrically simple lakes a single sampling location may be sufficient to adequately characterize lake. In morphometrically complex lakes, many sampling locations will be necessary to perform such characterizations and the procedures for data analysis become important as well.

Environmental problems that are acute or which only occur during certain seasons may require one or a few repeated sampling efforts. However, trends (especially those over long times) require regular repeated efforts. These time-series efforts can lend themselves to specialized statistical techniques associated with time series analysis. In those analyses the ability to resolve certain frequencies of phenomena depend not only on repeated observations but also on the frequency with which those observations are made. In plain terms, for example, diel processes require more than one or perhaps two observations per day, and perhaps many more if the process in question occurs rapidly at a particular time during the day (such as the behavior of schooling fish at dawn). There are still many lake ecological phenomena that have not been adequately resolved for lack of adequate sampling.

There are powerful new techniques for rapid sampling for special applications. For example, the incorporation of real-time GPS positioning with ADCP profiling or with rapid *in situ* methods provides high-resolution capability for spatial sampling over both short and long distances. Likewise satellite and aircraft-based sensor technologies can provide detailed characterizations of lake surface characteristics when they are employed with adequate field ground-truth measurements.

Such approaches are exciting but still limited in application. Remote sensing is generally accepted but only for lake surface characterizations. In as much as some correlation can be made to characteristics at depth, such surface characterizations can be used to provide broad, accurate descriptions of spatially complex lakes. However, such approaches provide only short-term (snapshot) characterizations and longer-term work often employs remote continuous monitoring techniques.

Remote continuous monitoring, in turn, depends on adequate maintenance and calibration of the sensor arrays. Such arrays can be limited to single devices or they can extend to large numbers of recording devices. The amounts of data produced by all of these technologically advanced approaches are often enormous and can tax data management hardware, software, and personnel time.

2.3.3.1 Temporal Dependence

The changes occur in lakes through the year demand different observations. For example, seasonal stratification demands intensive observation at depths where density changes most rapidly. This is usually in the region of the thermocline. In shallow, well-mixed lakes sampling would not need to be as intense. Exceptional situations, however, exist where thermal patterns and chemical patterns do not coincide. These situations are more common than not and parameters such as dissolved oxygen concentrations or specific conductance are useful indicators of such chemical trends. In some situations, pH is also useful. Adequate sampling in these situations requires either prior knowledge of expected chemical trends, application of field judgement based on measured in situ trends, or repeated sampling after laboratory analysis has identified depth ranges in which chemical trends are important. In the best studies all of these work together to formulate the most rational sampling effort.

Where inflows have periodic or sporadic changes, loading estimates may require intensive sampling of short-term events. Two common situations exist: upstream dams have releases dependent on water needs and power demands, and streams flowing in from the watershed experience base flows punctuated by storm high flow events. These two situations are quite different and require different strategies for sampling. Dam inflows are somewhat predictable and at most times the magnitudes of flow are known. Sampling these is a simple matter of coordinating sampling effort with expected operation.

Streams pose different problems. Where active gaging stations exist, records of flow are available and high flows can be anticipated on the basis of meteorological information. Sampling is problematic and involves either labor-intensive manual sampling or expensive mechanized sampling. Either approach can be expensive and can risk missing the events of interest.

During base flow conditions, sampling can be much less demanding. Samples merely need to be collected at dependable intervals and coordinated with the flow data. Upstream dams usually contribute their inflows almost solely during operation. Non-operational releases are not desired and measures are usually taken to minimize them. Therefore sampling is best during operation and flow data is usually immediately available.

Where flow data cannot be provided through other sources, inflowing streams must be rated and gaged on a temporary basis. This is a significant effort and requires mapping the stream cross-section according to shape and current velocity under different stage conditions. An equation can thereby be derived to predict flows at various stages. Once established, a simple staff gage can be employed during sampling. The authoritative source for this procedure and for most stream measurements is the U.S.G.S.

2.3.3.2 Loading

Once sampling is completed successfully, the concentration data may be combined with the flow data to produce flow-weighted quantities, average concentrations, etc. Such metrics for all inputs can then be combined to predict the loading to a lake. At the same time, outflows treated in the same manner can be subtracted from the loading estimate. The process is as simple as balancing a bank account and can be used to imply the loss or accumulation of materials to or from the lake.

2.3.3.3 Data Management Considerations

Obviously these assessment efforts proceed best with large quantities of data. Either these are from pre-existing sources or they are collected during the assessment studies. Data management must contend with a large number of issues all of which are important to the successful assessment.

Data will be available from different sources, in different forms, and in different quantities. Continuous thermal monitors may quickly and easily produce megabytes of thermal data in a string format. At the other extreme, certain biological analyses involving enzymatic reactions or isotopes may produce just a few estimates, the results of enormous effort. Operational data from a dam inflow may be available on printouts or in digital spreadsheet format. Decisions on how to manage these diverse sources and sizes must be made substantially before the data arrive. These should be available to all parties to the studies which requires that either all parties employ identical hardware and software...or else the data must be in a form that can be imported universally. Fortunately there are ways to accomplish this.

Today, data management requires computer assistance. There are three basic computational approaches to data management. First, all data may reside in a comprehensive file managed using a sophisticated program such as SAS (Statistical Analysis System) or SPSS. These software packages offer versatile capabilities of being able to manage a database, plot the data graphically, analyze it statistically, and this is accomplished through a programming language which enables programmed repetition of processes needed on a periodic basis.

The second approach employs a database program designed solely for that purpose and having few capabilities beyond simple manipulations of the database. Their power consists of an ability to move, sort, and query large bodies of data. Examples of these programs include Paradox and Access. Their advantage is their small size compared with the full-featured packages in the first approach. Their disadvantage is that they are limited and cannot provide more than rudimentary graphics or analysis.

A third approach is to manage data as separate files which contain similar or compatible information. For example, one file would contain in situ data collected from the lakes, another would contain stream data, and yet another would contain meteorological data or operational data. This third approach often tends to evolve as new sources of data begin to be collected during existing studies. In this approach the management tools can include statistics packages, database management software, or popular spreadsheet programs such as Excel. The advantage is that the management tool can be chosen to optimally fit the data needs. The disadvantages are that the data files exist in different formats that may not be compatible and that multiple files increase the possibility of management errors. The first disadvantage is easily overcome by maintaining files in an ASCII (text) format or by exporting them as needed in compatible formats. The second disadvantage is unavoidable.

A final mention of a recent development is necessary - GIS. Geographical Information Systems are important for management and analysis of spatially dependent data of many types. These systems are powerful tools but they are specialized in application. As methods such as remote sensing become more prevalent and the datasets produced become more common, GIS will be a dominant means of managing and understanding the results. However, for smaller datasets and those which are not spatially complex, GIS is not necessary and analysis can be completed with some attention to detail using other more conventional methods.

2.3.4 FIELD DATA - DATABASE

This step is important and is similar for the transition from the laboratory to the database as well. If field data are taken electronically, the transition is relatively simple and the manager merely must be careful that only the intended data are included in the database. The manager must also take care that all of the intended data are included. If taken manually, however, the paper datasheets must be transcribed into digital form. This is labor- and time-intensive. Once transcribed, a third party must proof-read the digitally printed form to detect transcription errors. Errors detected must be corrected and the new printout proof-read again. This process must be repeated until no errors are detected. This does not insure detection of all errors and the analysis stage serves as a final detection process. Identical precautions must be taken for laboratory or other sources of data such as old printed reports.

2.3.5 DATA ANALYSIS

There are many approaches to data analysis. However, all persons engaged in data analysis need to recognize that we live in exciting times - with new and greater computational ability each year. Everyone must remember, however, that computational tools are intended to aid in the analysis and are not ends in themselves. People create the ideas and people must judge value and results. There is still good work that can be done with modest computational means as long as there is sound thought.

If studies are designed according to strict statistical guidelines, then the first analyses are predetermined statistical routines. In reality, many assessment efforts do not have this luxury and this section will emphasize an approach which makes no such statistical assumptions and is called ‘exploratory data analysis’.

Exploratory data analysis is the approach to analysis popularized by John W. Tukey (1977) and his students. In this approach the assumption is that data is available and there may be no design whatsoever to its collection. The task of exploratory data analysis is to find trends or statistical relationships as efficiently as possible and with few preconceived assumptions.

Because this approach employs well-known accepted statistical procedures and exists mainly as a point of view, there are two ways to employ the approach just as in other statistical endeavors. The first way is to identify trends and use these to form predictive statistical relationships (models). The second way is to view the procedure as a means of discovering possible relationships from which to form new hypotheses for future studies. The two ways of employing this approach differ mostly in intent. The power of the approach is that it allows the most complete analysis of data.

The tools for data analysis begin with examination of the data. This is sometimes facilitated with tables of summary statistics but most often by simple graphs or scatter plots. Here the analyst can visibly detect potential trends. Repetition of the plots employing subsets or novel sorts of the data can bring out many potential relationships. Statistics then is ready to supply confidence to them or to help locate hidden relationships.

The tools required for statistical and graphical analysis are often different. However, the first approach to data management discussed here listed two software tools capable of doing nearly everything. Software packages designed specifically for statistical analysis include, for example, Systat which is a comprehensive statistics package capable of graphical representation as well. One package, Data Desk, was written specifically for exploratory data analysis. Graphical display can be accomplished using many off-the-shelf software packages such as SigmaPlot and DeltaGraph. Graphical software such as these have fuller capability than available with spreadsheet programs. Isoplots (three-dimensional graphics) may be created by DeltaGraph and some other very powerful programs such as Surfer and Spyglass. All of these programs continue to add capability and power.

2.3.5.1 Specialized Analyses

Data analysis can pursue simple exploratory methods or it can employ methods with specific goals. The data can be used to provide input to a variety of computational techniques ranging from tools such as SELECT to simple models such as PROFILE or BATHTUB to complex models such as CEQUAL and its many manifestations.

In one view all of these tools and models may be compared in terms of their simplicity, their ease of use, their generality, precision, and their realism. As often stated, all three of the last qualities are not often found together. The quest for realism has led to very sophisticated approaches such as

CEQUAL. The computational nature of these approaches makes them very precise. Consequently, their general application is difficult and each application must ensure that all assumptions and demands such as boundary conditions are thoroughly addressed. A precise, realistic result is not undesirable and the ability to achieve such results is a function of the proper choice of a tool for a particular question as well as proper application.

Computational tools such as SELECT are useful and can be applied to specific unsophisticated situations to answer basic questions. SELECT, for example, is capable of making predictions regarding the withdrawal zone in a lake and the quality of the water being released. It employs lake profile data and knowledge of the release structure to calculate the mix, if any, and the characteristics of release water to be expected. It is not a dynamic model, however, and makes no use of inflows, dynamic lake processes, or other factors that may prevent steady-state conditions. In this sense, this tool lacks elements of realism necessary for more sophisticated predictions.

More sophisticated tools such as PROFILE allow even further derivations from field data. PROFILE, for example, employs time series of lake profile data to calculate, among other things, oxygen depletion rates for depth ranges of interest. BATHTUB can provide much insight to loading trends with relative ease. However, it does not include detail on specific limnological processes that control or influence loading or which determine the sources or fates of loaded materials.

For such detail, numerical methods include very sophisticated models that do include such processes. However, models such as CEQUAL are far from trivial to use and the many types of input necessary for the best predictions are also sources of potential error. They do work, however, and when applied in a robust manner (results confirmation from several sources) they are powerful predictive and investigative tools.

2.3.5.2 Other Specialized Assessment Methods

There are numerous other ways to address lake issues. These are often related to specific questions and involve detailed specific studies. The assessment of dissolved oxygen dynamics and contributing factors is one example of these. A basic approach to field or laboratory measurement may be modified to fit the study needs in such cases. Long-term BOD measurements, sediment exchange measurements, etc. are techniques that cannot easily be applied in a routine manner. However, it is very informative to have the results, say, of long-term sediment oxygen demand if other methods have not provided needed understanding of the oxygen dynamics.

In each of these cases, the critical element is trained personnel with good understanding of the aquatic system. This facilitates understanding of the results and allows identification of additional needs when conventional assessment methods fail.

2.3.6 CASE STUDY - RICHARD B. RUSSELL LAKE

This case study combines nearly every element of assessment mentioned in this section. Completed in 1983, Richard B. Russell Lake (RBR) impounded the Savannah River downstream from Lake Hartwell (HW) and upstream from J. Strom Thurmond Lake (JST). Studies were initiated by USACEWES sponsored by USACESAS in order to examine the early development of the new lake. Historical data was available and this was added to the new database formed of *in situ* measurements and laboratory analyses. In addition, biological studies of many kinds were initiated by several cooperating agencies and universities. The database was managed on a minicomputer using SAS. As the desktop computer gained power, this management migrated to the desktop and many of the programs mentioned here were used to examine the results of these studies. In addition, the agencies and universities employed incompatible platforms and software and methods of compatibility had to be discovered.

Throughout the project and to the present time there has been a continuous effort to monitor outflows for the purpose of water quality description. In addition, a large-scale lake oxygenation system at RBR is managed using the data produced by the continuous monitoring effort as well as the lake studies.

Specialized studies of sediment deposition provided important insight to the processes in which iron and manganese were released or deposited under certain conditions in RBR and JST and how these processes were influenced by oxygenation in RBR. Other studies of inflows provided model input for loading estimates. ADCP surveys (water current profiling) provided knowledge of lake and outflow water movement patterns under different conditions. These surveys also provided comparison to physical model predictions for specialized situations.

The construction project continued into the 1990's and by that time, the database outgrew the ability to easily manage it from the desktop. By this time, studies of loading, in-lake oxygenation, lake aging, productivity, zooplankton, phytoplankton, fisheries, etc. had been added as datasets. What had been originally conceived within a single comprehensive structure had necessarily evolved to numerous databases, some very large. Continuous monitors and *in situ* logging created 100's of megabytes of data while conventional profiling contributed more modest datasets. A new construction project (pumped storage) was initiated in the 1990's. The field facility incorporated a Novell network (fiber optic base) and ultimately maintained two servers primarily for data management. Gigabytes of data were tracked and managed. Smaller portions (<650 megabytes) were disseminated or archived on CDROM's. And the main database was moved to a workstation at WES and is still manipulated with SAS. However, the analyses are mostly addressed using desktop applications and graphic software, some of which were listed earlier.

The final phase of assessment of the effects of large-scale pumped storage in the presence of upstream and downstream reservoirs and in-lake oxygenation employed every aspect of these studies.

These also contributed to a modeling effort using CEQUAL to dynamically link RBR and JST in the presence of pumped storage. This effort is coming to fruition with the ability to accurately predict thermal and dissolved oxygen and trends based on different limnological and operational conditions.

The predictions of ecological effects of pumped storage were employed in decisions on how to minimize impacts using modified structures and operations. Once in commercial operation, the final phase of study will be to confirm the predictions through follow-up monitoring of critical limnological characteristics.

2.3.7 REFERENCES

Environmental Protection Agency. 1979. "Methods of Chemical Analysis of Water and Wastes," Environmental Monitoring and Support Laboratory, USEPA, Cincinnati, OH.

Gaugush, R. F.   1987. "Sampling Design for Reservoir Water Quality Investigations," Instruction Report E-87-1, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.
Gaugush, R. F. 1993. "Sampling Design Software User's Manual," Instruction Report W-93-1, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Gilbert, R. O. 1987. *Statistical Methods for Environmental Pollution Monitoring*, Van Nostrand Reinhold, New York, NY.

Green, R. H. 1979. *Sampling Design and Statistical Methods for Environmental Biologists*, John Wiley and Sons, New York.

Greenberg, A. (Ed.) 1995. *Standard Methods for the Examination of Water and Wastewater*, 19th Edition, American Public Health Association, American Water Works Association, Water Pollution Control Federation. Washington, D.C.

Montgomery, D. C. 1997. *Design and Analysis of Experiments*, Fourth Edition, John Wiley and Sons, New York.

Neter, J. And W. Wasserman. 1974. *Applied Linear Statistical Models*, Richard D. Irwin, Inc. Homewood, IL.

Olem, H. And G. Flock, eds. 1990. "Lake and Reservoir Restoration Guidance Manual, 2nd edition," EPA 440/4-90-006. Prep. by N. Am. Lake Manage. Soc. For U.S.Environ. Prot. Agency, Washington, D.C.

Principles and Procedures of Statistics, 3rd ed., by Steele, Torrie, & Dickey

Reckhow, K. 1979. "Quantitative Tools for Trophic Assessment," Spec. Rep. U.S. Environ. Prot. Agency. Washington, D.C.

Sokal, R. R. and F. J. Rohlf. 1969. *Biometry*, W.H. Freeman and Company, San Francisco.

Stauffer, R. E. 1981. "Sampling Strategies for Estimating the Magnitude and Importance of Internal Phosphorus Supplies in Lakes," EPA-600/3-81-015. Corvallis Environ. Res. Lab. U.S. Environ. Prot. Agency. Washington, D.C.

Steele, R. G. D. and J. H. Torrie. 1980. *Principles and procedures of statistics: a biometrical approach*. McGraw-Hill. New York

Strickland, J. D. H. And T. R. Parsons. 1972. *A Practical Handbook of Seawater Analysis*, Fisheries Research Board of Canada. Ottawa.

Thomann, R. V. and J. A. Mueller. 1987. *Principles of Surface Water Quality Modeling and Control*, Harper and Row, Publishers, Inc. New York, NY

Tukey, J. W. 1977. *Exploratory Data Analysis*, Addison-Wesley, Reading, MA.

Wetzel, R. G. And G. E. Likens. 1991. *Limnological Analyses*, 2nd ed. Springer Verlag. NY

2.4 ASSESSMENT OF TAILWATER PROCESSES

2.4.1 INTRODUCTION

The assessment of water quality processes that suggest enhancement techniques may be applicable in a reservoir tailwater, begins with a clear definition of the perceived problem(s) and identification of the objectives to be accomplished. Once the problem(s) has been determined and objectives defined, investigations of processes likely contributing to the problems can be conducted. These processes may be occurring in the watershed, the reservoir, or the tailwater and may be associated with reservoir operations, natural conditions, or existing conditions. Determination of the contributing processes leads to identification of variables to measure (e.g., hydrology and other physical features, water quality constituents, and biota). These processes occur over a variety of time scales (e.g., microbial life cycles are measured at hourly or daily scales compared with years for fish, water movement can be measured in seconds (velocity), days to months (flood attenuation), or years (repetitive cycles or high/low annual flow)). These varied time scales are associated with data collection needs for meeting objectives and, in part, determine the type of equipment necessary for successful data collection.

The next step is the careful and thoughtful design of the sampling program with consideration given to cost constraints, logistics, scientific soundness (including quality assurance and quality control), and meeting stated objectives (if possible). Implementation almost always requires some trial and error and readjustment so allowing for site visits, trial studies, and periodic review should be incorporated into sample design. During and after sampling, data analysis (beginning with a good database management plan) should be an ongoing activity. Attention should be given to equipment and analytical calibrations, accuracy, and precision, accurate data entry and conversions, and the integrity of databases during transport to graphic and analytical packages. The final step involves combining the above activities with available tools, such as models, for developing an approach or technique for enhancement. Knowledge of available tools and required inputs should be used in the previous steps. Coordination with others should also be a component of the assessment and some ideas are presented in a later section.

The objective of this section is to review the assessment process for identification of problems, formulating solutions, and implementing enhancement techniques. Problem identification is presented in a previous section and there are numerous references for sample design and data analysis (e.g., Montgomery 1997; Gilbert 1987; Green 1979). Emphasis will be placed on sampling considerations and analytical tools.

2.4.2 SAMPLING CONSIDERATIONS

Problem identification in reservoir tailwaters usually occurs via a downstream user (e.g., water supply, recreational users) or by personnel in close proximity to the project. For example, fish kills or noxious odors from the degassing of hydrogen sulfide are usually reported by fishermen, people at tailwater recreation sites, or project personnel. High concentrations of iron or manganese cause increased treatment costs for potable water suppliers who routinely monitor these constituents and often report unusually high concentrations. Biological monitoring by state and federal agencies may detect long term trends in biota (e.g., changes in community composition, irregular growth). Other forms of problem identification include knowledge of episodic events such as chemical spills, noxious algal blooms, or routine water quality monitoring (e.g., measurements of temperature or dissolved oxygen concentrations).

Once the problem has been identified, objectives must be determined, clearly stated, coordinated with other involved or interested parties, and evaluated for attainability. For example, hydrogen sulfide odors generated by the degassing of hydrogen sulfide are obnoxious and potentially harmful. The odors are indicative of severe anoxia in the upstream reservoir and a clear objective would be to prevent the occurrence. However, in-lake measurements to avoid the development of anoxia may not be plausible or could be extremely expensive. The objective to prevent the occurrence may not be attainable since the enhancement technique may not be applicable.

Processes contributing to the identified problem must be determined and evaluated. Often several processes are involved and an approach that considers interactions between processes is required. The role of the hydrology and morphology of the tailwater must be determined. Often a problem such as low dissolved oxygen concentrations is more pronounced during low flow (especially at night) than at high flow. Conversely, resuspension of sediments is greater during high flow (particularly on the leading edge of the hydrograph). This type of information assists in sample design, identification of constituents of interest, and evaluation of applicable enhancement techniques.

Constituents of concern (excluding variable flow) most often include temperature, dissolved oxygen, nutrients, metals, particulates, and algae. Most often the severity of water quality problems associated with these constituents is related to the time scale or duration of abrupt or lasting changes. For example, a sudden increase in water temperature is an acute impact on biota whereas a slower increase in water temperature (e.g., seasonal warming) may have no impact or a chronic impact (e.g., timing of increased temperatures coincides with spawning seasons and too high temperatures inhibit spawning or decrease survival rates). As described earlier, dissolved oxygen concentrations in tailwaters follow daily and seasonal patterns and seasonal trends in nutrients and metals are often apparent. Knowledge of these patterns is necessary for the efficient assessment of water quality problems (sample design, data analysis, etc.).

Another important consideration is the application of remote monitoring and discrete sampling for assessing water quality. Continuous monitoring of temperature, dissolved oxygen, pH, and specific conductance in situ is the most common method of remote monitoring. Sensors for light, oxidation-reduction potential, ammonia, and turbidity (and other constituents) are also available. Vorwerk et al. (1996a) have evaluated some of these methods for monitoring releases from hydropower projects and their results are included as Appendix 2.4A. Although these studies were conducted at hydropower projects, concepts of spatial and temporal heterogeneities, sampling logistics, and data handling are easily applicable to tailwaters and at locks and dams. Methods for the design and installation of a remote monitoring system have been described by Lemons et al. (1998) and are included as Appendix 2.4B. Guidelines for implementing a monitoring program with remote monitors are presented in Figure 2.4.1

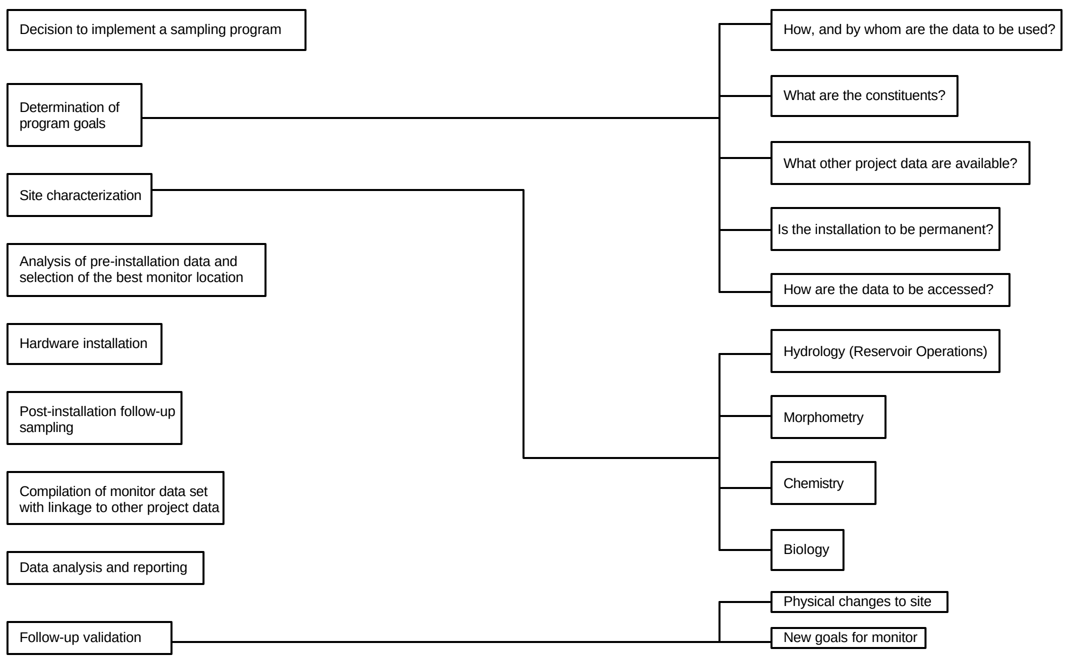


Figure 2.4.1 Decision diagram for implementing a monitoring program with remote monitors. Modified from Lemons et al. (1998).

The use of remote monitors results in the development of a large database of time-series data that provides a lot of information. The transfer of the data from a collection site to a remote user is accomplished via software and modems (Figure 2.4.2). Often data interpretation is made easier with the use of simple statistics such as ranges and mean values. These calculations can also be used for assessment of equipment calibrations (ranges above actual real environmental ranges which would indicate calibration/maintenance problems) and undesirable water quality conditions. Vorwerk et al. (1996b) address the issue of statistical verification of data from remote monitors in a study included as Appendix 2.4C. This study emphasizes the need for determining heterogeneities and establishing the representativeness of data collected from a fixed site. Statistical techniques for accomplishing these tasks are presented.

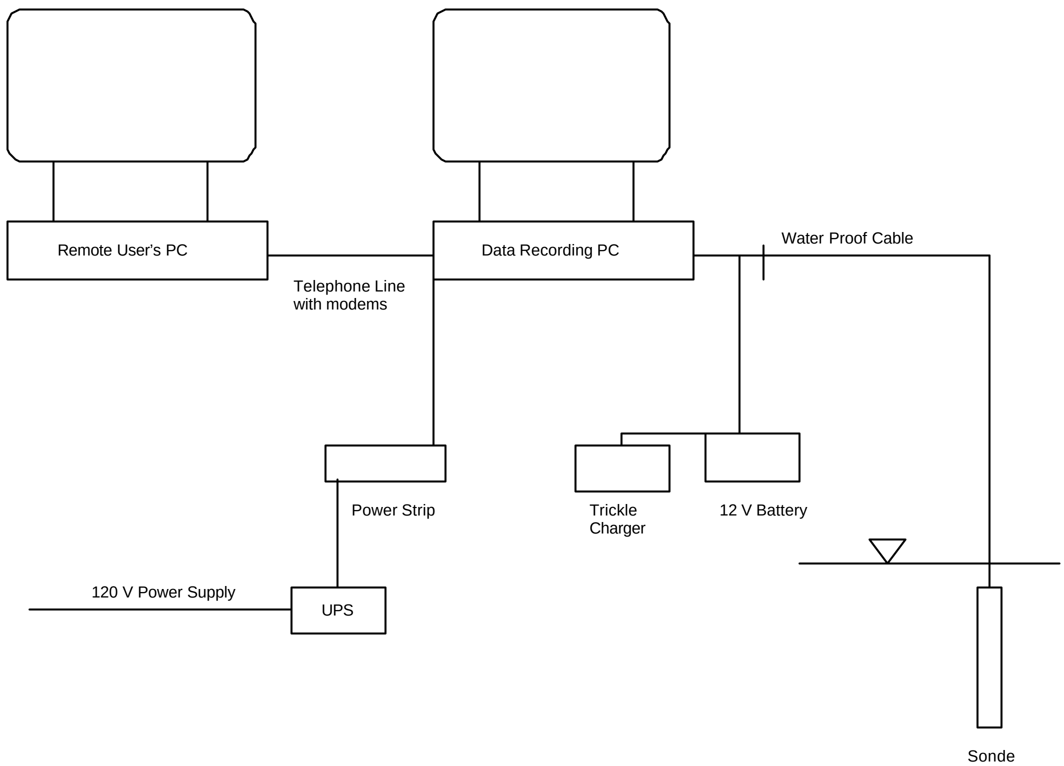


Figure 2.4.2 Schematic of monitor setup and data transfer to a remote user

2.4.3 ANALYTICAL TOOLS

Hydrology is often described using stream gage techniques and reservoir operation records as described in Chapter 1.3. Information about the channel morphology is often obtained when channel discharge measurements are made but are applicable only to the transects established. Information about substrate type and morpholgy between transects is lacking. One method of providing some of the missing information is to survey the river reach of interest in more detail and collect substrate samples for analysis. The level of detail of characterizing the study reach is determined by the objectives of the study and available resources. Substrate sampling is often quite difficult if the substrate sizes vary from boulders to fine sediments. In areas with comparable substrates, samplers such as Ponar, Ekman, or clamshell dredges are most commonly used. In areas of deposition, sediment cores may be collected. In areas with large cobble an appropriately sized quadrant may be used to define the perimeter of the sample area. Often several sites are randomly selected at a particular site and the samples are either analyzed separately for statistical comparisons or composited. Additional sources of hydrology and morphology may be available at the project office, in the appropriate section of the managing agency, or from previous studies conducted by university personnel.

Techniques for sampling the biological community are often specific for the biota of interest. For example, hydroacoustic techniques, electroshocking, and poisoning are useful for studies of fish communities while deployment of divers with quadrants is more applicable to studies of benthic communities such as mussel beds. Studies of invertebrates, such as aquatic insects, often requires several techniques utilizing substrate sampling, a variety of nets and collectors, and deployment of artificial substrates. Certainly, clearly stated objectives in assessing tailwater habitats are essential. Some techniques for evaluating aquatic habitats in rivers (applicable to tailwaters) and associated data analysis and interpretation can be found in Sanders (1991). Some considerations for fisheries in reservoir releases are presented in Nestler et al. (1986).

2.4.4 REFERENCES

Gilbert, R. O. 1987. *Statistical Methods for Environmental Pollution Monitoring*, Van Nostrand Reinhold, New York, NY.

Green, R. H. 1979. *Sampling Design and Statistical Methods for Environmental Biologists*, John Wiley and Sons, New York.

Lemons, J. W., Vorwerk, M.C., Carroll, J. H ., and Jabour, W. E. 1998. "Remote Monitoring of Hydroprojects: Design, Installation, and Verification of Remote Monitoring Systems," Water Quality Technical Note AM-02, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Montgomery, D. C. 1997. *Design and Analysis of Experiments*, Fourth Edition, John Wiley and Sons, New York.

Nestler, J. M., Novotny, J. F., Jacobs, K. E., and Swink, W. D. 1986. "Handbook on Reservoir Releases for Fisheries and Environmental Quality," Instruction Report E-86-3, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Sanders, L. G. 1991. "Techniques for Evaluating Aquatic Habitats in Rivers, Streams, and Reservoirs; Proceedings of a Workshop," Miscellaneous Paper W-91-2, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Vorwerk, M. C., Jabour, W. E., and Carroll, J. H. 1996a. "Evaluation Methods for In Situ Monitoring of Releases from Hydropower Projects," Water Quality Technote AM-01, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Vorwerk, M. C., Carroll, J., and Lemons, J. 1996b. "Statistical Verification of Mean Fixed Water Quality Monitor Sites in Flowing Waters," Water Quality Technical Note AM-03, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

CHAPTER 3

INTERACTIONS WITH OTHERS

INTRODUCTION

This chapter is designed as a working chapter with input from workshop participants. Background information describing coordination processes such as defining management objectives and goals, roles of local governments, landowners, and other sectors, financial considerations such as leveraging and partnering, developing material for public education, and interagency cooperation is provided for discussion and expansion based on the experiences and comments of all participants. Topics listed in the discussion section at the end of this session reflect comments from participants at previously conducted workshops. Additional information is provided in the Clean Water Action Plan: Restoring and Protecting America's Waters available from the U.S. Government Printing Office (ISBN 0-16-049536-9) and at the website:

http://www.cleanwater.gov

3.1 COORDINATION

As interest in improving water quality develops, whether it is from the private sector, in response to acute or chronic problems, or is mandated by regulations, coordination is necessary in all phases of any attempts to ameliorate problems. While understanding watershed, reservoir, and tailwater processes, assessing these processes, and defining the problem(s) are also necessary, proper coordination as early as possible will be the most efficient method to achieving improvements. The people involved (those with vested interests in or responsibilities for the resources, i.e., the stakeholders) must be identified and work together to develop a plan to study the problem and implement recommendations in a manner that is acceptable to all if possible. This often requires some compromises, distribution of responsibilities, and pooling of resources (e.g., expertise, equipment, funds). Local investment and coordination is extremely useful. Outside experts can help (or hinder) a project, but it seems as though some good results are obtained when local people provide much of the leadership. Management objectives must be clearly defined, roles and support from stakeholders must be determined, education for and input from the public must occur, and interagency cooperation must be established at technical and managerial levels. Examples of this process include the coordinating activities for the Comprehensive Basin Study of the Alabama-Coosa-Tallapoosa/Apalachicola-Chattahoochee-Flint River System and Coffey et al., 1992 (Appendix 3.1A). Since this process will vary by region, this section will attempt to define in a general way an approach to developing the coordination necessary, provide some rationale for the steps suggested for coordination, and provide the workshop participants with enough insight to develop or assist in the development of any studies they may become involved with in their region.

3.1.1 DEFINING MANAGEMENT OBJECTIVES

Once a water quality problem has been identified and appropriate processes assessed, clear and realistic management objectives must be established. Previously conducted assessments were probably conducted with a “rough” idea of what water quality improvements (management objectives) were desired. Items to consider include but are not limited to:

1. Who are the stakeholders - involved agencies, users, etc.?
2. What are the roles of the stakeholders?
3. What are the water uses and are there potential conflicts?
4. How will objectives be determined?
5. Others?

Perhaps, in addition to objectives, one or more measurable goals could be established (e.g., restoration of a fishery, reduction in algal blooms, increased dissolved oxygen in the tailwater, etc.). Developing time lines, funding sources, and identifying who can participate are also part of defining objectives and developing a management plan. Table 3.1.1 provides an example of management topics such as the program purpose, contract period, and eligibility for conservation programs for watershed management. Examples of this approach for nonpoint source pollution control are provided as Appendices 3.1B and 3.1C. Consideration of these items can result in the development of a management team and plan and provide for an efficient approach to solving the water quality problem.

3.1.2 STAKEHOLDERS AND THEIR ROLES

As previously defined, stakeholders are those individuals, agencies, and water users with a vested interest in or responsibility for the water resource. Many of these interests and responsibilities are linked and are often competing and stakeholders are often required to develop a management plan that “protects” their interests.

3.1.3 FINANCIAL CONSIDERATIONS

What does it cost and who is going to pay? Determining costs for water quality management often includes consideration of not only capital expenditures, but services provided as well. These services are often substantial and can be used to “partner” or cost-share with federal dollars. Interactions with local, state, and federal agencies is imperative to combine funding sources to accomplish management techniques. A list of federal agencies, their programs, and funding are provided in Table 3.1.1.

Table 3.1.1 Clean Water and Watershed Restoration Budget Initiative		
FUNDING BY AGENCY	**1998 (Enacted) $ in millions**	**1999 (Budget) $ in millions**
ENVIRONMENTAL PROTECTION AGENCY		
State Grant Assistance		
Polluted runoff control grants (Section 319)	105	200
State program management grants (Section 106)	96	116
Wetlands protection grants	15	15
Water quality cooperative agreements	20	19
Water quality program management	248	279
DEPARTMENT OF AGRICULTURE		
Natural Resources Conservation Service: Environmental Quality Incentives Program	200	300
Natural Resources Conservation Service: Locally led conservation	0	20
Natural Resources Conservation Service: Watershed health monitoring	0	3
Forest Service: Improve water quality on federal lands	239	308
Agriculture Research Service: Watershed health research	0	2
DEPARTMENT OF THE INTERIOR		
Bureau of Land Management: Improve water quality on federal lands	133	157
Office of Surface Mining: Clean streams	5	7
US Geological Survey: Water monitoring and assessment	125	147
Fish and Wildlife Service: Wetlands restoration	36	42
Bureau of Indian Affairs: Improve water quality on tribal lands	0	5
NATIONAL OCEANOGRAPHIC AND ATMOSPHERIC ADMINISTRATION		
Polluted runoff and toxic contaminants	0	13
Harmful algal blooms	0	9
ARMY CORPS OF ENGINEERS		
Wetlands program	106	117
Challenge 21: Floodplain restoration initiative	0	25
INTERAGENCY PROJECTS		
Florida Everglades	228	282
California Bay Delta	85	143
Elimination of overlap between Everglades and other water programs listed above	-5	-5
TOTAL CLEAN WATER AND WATERSHED RESTORATION INITIATIVE	**1,636**	**2,204**

Various sections of the **Clean Water Act** provide funding for water quality management efforts. **Section 305(b)** establishes a process for reporting information about the quality of the Nation's water resources. **Section 314** of the Clean Lakes Program, historically, has awarded grants for the study and restoration of publicly owned lakes. The 1987 Water Quality Act Amendments to the Clean Water Act added **Section 319**, which establishes a national program to control nonpoint source pollution. **Sections 303 and 304** of the Clean Water Act require states to protect the biological integrity as part of their water quality standards. **Section 303(D**) of the Clean Water Act requires each state to establish, in accordance with its priority rankings, the total maximum daily load (**TMDL**) for each waterbody or reach identified by the state as failing to meet or not expected to meet water quality standards after imposition of technology based controls. **Section 402** of the Clean Water Act provides authority for issuing National Pollutant Discharge Elimination System (**NPDES**) permits. Collaborative efforts for watershed protection are also possible through the Federal Interagency Unified Watershed Assessments (**UWA**) Action Team. This approach is being evaluated and additional information is available from Barry Burgan at 202-260-7060.

3.1.4 RISK ASSESSMENT

Ecological risk assessment is defined as "The process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors" (USEPA 1992). Risk management is a decision-making process that involves all the human-health and ecological assessment results, considered with political, legal, economic, and ethical values, to develop and enforce environmental standards, criteria, and regulations (Maughan 1993). This process can be applied to a specific site or based at a geographic scale such as a river reach or watershed.

3.1.5 PUBLIC EDUCATION

While it is important to define a water quality problem based on technical studies or perceptions, educating the public or community about the problem, why it occurs, who it effects, what can be done about it, and what will it take to do something about it is also important if an enhancement technique is to be applied. Local programs administered by District Conservationists and Extension Agents are excellent starting points for public education. These individuals usually know the local stakeholders quite well and often have developed (or can develop) good working relationships. A lot of literature designed for public education is available through these resources. Additional information is available at the state and federal level via water resource publications such as state sponsored publications (e.g., NWQEP NOTES, The NCSU Water Quality Group Newsletter) and federal publications such as The Water Monitor and News-Notes (EPA). Numerous web sites are also available and a good starting place is:

http://www.epa.gov/OWOW/

3.1.6 DISCUSSION

General topics considered important by previous participants for further discussion:

1. Educating the community is considered to be extremely important. People need to be informed of problems, why the problems exist, what can be done about the problem, why the problem should be fixed, and what will it take (time, money, resources) to fix the problem.

2. Financial coordination is imperative since in most cases funds and resources will be required from many sources. Diverse funding sources are necessary.

3. Management efforts are often influenced by time constraints such as required spending in a fiscal year. More flexible use of finances should be available to implement management techniques. The funding should fit the implementation not the reverse.

4. Targeting funding to areas that contribute the most to the problem or that may respond best to management techniques should be incorporated into planning processes.

5. Preventive investment should be supported at a higher rate/level.

6. Volunteer monitoring, media coverage, and project visibility help create environmental responsibility and a sense of ownership that heightens awareness of problems and solutions.

7. Compare state programs and approaches. Encourage interstate exchange of ideas, especially with more successful programs.

8. <u>Cost-effectiveness Analysis</u> by Levine (published by Sage publications) might come in handy when trying to put a dollar value on some of the unquantifiable things like a clean environment.

3.1.7 REFERENCES

Coffey, S. W., J. Spooner, D. E. Line, J. A. Gale, J. A. Arnold, D. L. Osmond, and F. J. Humenik. 1992. “Elements of a Model Program for Nonpoint Source Pollution Control,” National Water Quality Evaluation Project, North Carolina State University, Raleigh, NC.

Gale, J. A., D. L. Osmond, D. E. Line, J. Spooner, J. A. Arnold, G. D. Jennings, and F. J. Humenik. 1995. “Planning and Managing a Successful Nonpoint Source Pollution Control Project,” North Carolina Cooperative Extension Service, North Carolina State University, Raleigh, NC.

D. L. Osmond and J. A. Gale. 1995. "Farmer Participation in Solving the Nonpoint Source Pollution Problem," North Carolina Cooperative Extension Service, North Carolina State University, Raleigh, NC.

Maughan, J. T. 1993. Ecological assessment of hazardous waste sites. Van Nostrand Reinhold, New York, NY.

U.S. Environmental Protection Agency. 1992. Framework for ecological risk assessment. EPA 630-R-92-001. U.S. Environmental Protection Agency, Washington, DC.

CHAPTER 4

ENHANCEMENT TECHNIQUES

4.1. INTRODUCTION TO ENHANCEMENT TECHNIQUES

Water quality enhancement opportunities can generally be in the watershed, in the reservoir, and in the tailwater. To successfully identify an appropriate alternative, one must have a clear understanding of the processes causing the water quality problem and the processes that will improve or correct the problem. Hence, the previous chapters covering watershed, limnological, and operational processes. In most situations, to improve or resolve a water quality problem, will require the implementation of multiple alternatives, perhaps from all three areas of opportunity.

This chapter focuses on alternatives in these three areas. Watershed techniques are discussed that are related to agricultural and livestock, farming, and forestry practices and other human activities. In-reservoir techniques are presented to modify in-reservoir water quality or in-reservoir processes that improve water quality in the reservoir and in the tailwater. Also discussed are in-structure techniques that generally treat water during release to improve the quality downstream. Tailwater techniques are discussed that improve quality after the water has been released. Other miscellaneous enhancement opportunities are discussed.

The watershed techniques discussed include farming practices along streams, tillage versus no-till farming, strip-cropping, pesticide/herbicide control, livestock waste control, forestry/logging sedimentation control, and many other activities. In-reservoir enhancement techniques include selective withdrawal and several operational derivatives, aeration/oxygenation, destratification and localized mixing, and several operational modifications. In-structure techniques deal mostly with hydroturbine releases and aeration or oxygenation of release water during passage through the turbine. Tailwater techniques provisions for minimum flow and aerating weirs.

4.2 WATERSHED TECHNIQUES

The agricultural perspective of watershed management is multidisciplinary and multiagency in nature. Farm history, rural sociology, economics, planning, agronomy, soil science, and engineering are typical disciplines found in successful watershed water quality projects. Agency participants may include the Soil and Water Conservation District, the United States Department of Agriculture-Natural Resources Conservation Service (USDA-NRCS), Cooperative Extension Service, USDA-Farm Services Agency, and others. Agency representatives having experience in a total of many disciplines often work together to plan and implement a watershed project.

A one to two year pre-implementation planning period is needed to identify and document the water quality problem, develop objectives and goals, garner community and farmer support, and target critical areas for treatment (Gate et al, 1993). Pre-implementation planning may also include advance information and education programs, surveys to identify needs, and writing BMP implementation plans for farms already identified for treatment. Watershed planning may proceed as the result of a voluntary or legislative initiative.

4.2.1 TARGETING OR COMMAND AND CONTROL

A proven approach to achieving management objectives is to set priorities and direct resources to the greatest need. Land treatment must be targeted to critical areas or pollutant sources where the greatest improvement in water quality can be achieved for the least investment in BMPs (Maas et al. 1987). Defining critical areas involves identifying the greatest pollutant sources and assessing the hydrologic transport system, beginning at the water resource and moving upgradient to pollutant sources near major tributaries, lower order waterways, and then headwater streams. Land treatment can then be targeted by first estimating the magnitude of the pollution potential of the sources and then ranking them with respect to the quantity of pollutant(s) expected to affect the water resource. In this way, the resources of the project can be focused on the major sources of pollution, thereby increasing the probability of reducing pollutant exports and reversing the water quality impairment.

Develop scientific criteria for identifying critical areas and apply the criteria consistently. For watersheds with animal waste issues, evaluate the pollution potential of cattle lounging in streams, untreated animal operations, exposed manure piles, and land application areas. For sediment problems resulting from eroding cropland, evaluate erosion potential, existing BMPs, the distance of the source to the receiving water body, and the potential for pollutant trapping in the stream system. Pollutant transport models such as AGNPS (Young et al. 1995) may be used to develop scientific criteria to identify critical areas.

Command and control results when BMPs are mandated by the federal or state government. For example, the federal government has imposed the CAFO regulations for confined animal operations. In North Carolina, the Environmental Management Commission proposed rules in the summer of 1996 for reducing point source and nonpoint source loads of nitrogen to the Neuse River by 30 percent in five years. In order to achieve this 30 percent reduction, mandated solutions are proposed for different pollution sources. For agriculture, the following combination of BMPs may be required: 1) nutrient management and controlled drainage or 2) nutrient management or controlled drainage and a forested riparian buffer or vegetative filter, or 3) forested riparian buffers and vegetative filter strips when controlled drainage and nutrient management are not utilized (NCDWQ, 1996). These regulations proposed the reestablishment of riparian buffers along both sides of perennial and intermittent surface waters in the Neuse River basin, exclusive of ditches.

4.2.2 TECHNOLOGICAL DESCRIPTIONS

A BMP system is often required to effectively control NPS pollution, especially when the critical pollutant source involves an animal operation. Most animal operations require a combination of BMPs such as waste handling and manure application. Other BMPs may be needed to divert clean water away from the site as well as feedlot runoff control. Cropland BMP systems may include terraces and grassed waterways along with cover crops and conservation tillage.

4.2.2.1 Best Management Practice Systems

BMP systems that include a structure are very common for animal waste and cropland treatment. Structures include holding ponds, pits, lagoons, dry stacks, sediment basins, and detention ponds are often necessary to control pollution from farmsteads and fields; however, these structures do not achieve maximum effectiveness in isolation. To maximize the effectiveness of a structure, they must be combined with management practices such as waste utilization, nutrient management, or erosion control.

Nutrient and pesticide management are, in general, the most cost-effective BMPs in terms of requiring the least monetary investment for the greatest water quality benefit.

Sediment. Soil particles are detached by raindrop energy, overland flow, or wind energy. The erosion of soil particles is a natural geologic process, but it can become accelerated when the soil surface becomes exposed. Sediment loss can be reduced by BMPs that minimize exposure of the soil and soil particle detachment. Such practices include conservation crop rotations, cover crops, and conservation tillage. These BMPs improve soil structure and increase soil organic matter content and surface roughness. Where permanent gullies form in a field, a grassed waterway can be used to reduce gully erosion and provide a stable channel to divert runoff.

To minimize the transport of sediment there are several BMPs that reduce slope length and steepness or water velocity. Stripcropping, terraces, and contour plowing or row arrangement serves to reduce the slope length and reduce sediment transport. Field borders, filter strips, and level spreaders trap sediment in field runoff. Sediment basins and forest and grassed riparian buffers trap sediment before it reaches waterways.

Nutrients. Nutrient management includes matching nutrient application rates with crop needs, placing fertilizer correctly to optimize crop uptake, and timing fertilizer application to meet crop growth requirements. Sediment-attached nutrients can also be controlled by sediment control BMPs. Nutrients can also be intercepted or transformed by BMPs such as cover crops, riparian buffers, controlled drainage, or created in-stream wetlands. Cover crops can also adsorb residual nitrogen from deep in the soil profile, thus reducing nutrient losses. Nitrate may be removed in riparian buffers through denitrification and plant uptake, whereas organic nitrogen and phosphorus, attached to sediment may be reduced through sediment trapping practices.

Pesticides. Pesticides can be removed by using cover crop management and integrated pest management techniques of applying pesticides only when needed with the proper type applied, at the correct rate, and time. The transport of sediment attached pesticides can be slowed by the same type of BMPs that are used to trap sediment. Pesticides that are adsorbed to soil particles can be trapped in filter strip and riparian buffers.

4.2.2.2 Best Management Practices

The effect of a Best Management Practice (BMP) on the receiving water body is very site specific and is influenced by the extent of the problem, meteorology, hydrology, distance to the water body, treatment and maintenance (NWQEP 1989). In determining management options, consider that a reduction in erosion rate may or may not result tin improved water quality. Nutrients and chemicals in solution or attached to fine sediment particles can cause water quality problems independent of the volume of gross erosion. The amount of applied nutrients that reach a water body may be more of a function of runoff or fine sediment loss that total soil loss.

Best management practices reported in the literature typically differ by design and specifications from conservation practice standards in SCS field office technical guides. Research results from the literature are often edge-of-field studies with highly controlled conditions that may not relate directly to field conditions particularly due to problems of scale. Factors unrelated to experimental design that affect treatment strength include information and education, technical assistance, conservation ethic, economy, and government support. Table 8.3 is a general guide for BMPs discussed in this chapter.

Construction. A RMS for urbanizing land may include BMPs to control erosion on lands undergoing development for non-agricultural uses such as housing, industrial areas, recreational areas and roads. BMPs provide for water disposal and the protection of the soil surface. Smolen et al. (1988) developed a state-of-the-art design manual for planning and implementation of erosion and sedimentation control practices for disturbed areas in North Carolina. Approximate effectiveness for construction site erosion and sediment control BMPs are given by Brach (1989).

Compared to other BMPs, little is known on the effect of construction practices on nutrient runoff. Where excavation is extensive, nutrient exports from construction sites may be less of a problem than the nutrient export from other areas due to the limited nutrient content of many subsoils. Vegetative cover is very effective in keeping soil in place. Sediment fences, inlet protection devices, traps, and basins can retain up to 95% of large sediment particles.

Urban. Nutrient loading from urban areas is a growing concern. Best management practices to reduce nutrients in urban runoff include source, vegetative, and structural controls such as infiltration and detention devices. To maximize impacts, a combination of BMPs may be needed to achieve management goals.

Source Controls. Reducing the amount of nutrients applied to lawns, limiting the accumulation of leaf litter, proper nutrient storage, and diverting or excluding discharges from storm drains can control source nutrients. Other BMPs include land use restrictions (e.g. limiting impervious area, restricting development in riparian areas, and using grassed swales for drainage instead of curbs and gutters and piped drains. Source control effectiveness increases as the extent of treatment approaches the need.

Proper timing and rate of fertilizer application to lawns maximizes plant uptake and minimizes losses. Morton et al. (1988) found mean annual flow-weighted inorganic nitrogen concentrations in percolate were 0.36 mg/l for an unfertilized, over-watered control and 4.0 mg/l for the high nitrogen (244 kg N/ha-yr) application treatment. Annual losses were 2 kg/ha for the control and 32 kg/ha for the over-fertilized lawn. A study in Minnesota on the effect of keeping leaf litter and grass clippings free from street side curbs has shown phosphorus levels were reduced by 30 to 40% (Shapiro and Pfannkuch 1973). However, Athayde et al. (1983) found that street sweeping alone was not effective in reducing total phosphorus or total nitrogen concentrations in urban runoff.

Public education to increase the understanding the of problem and voluntary participation are generally required to implement urban source controls. Technical assistance may be needed for stabilizing sites with vegetation or for recommending fertilizer application rates. There are few studies that have monitored the effect of urban source area controls.

Vegetative Controls. Grassed swales, buffer strips and wetlands increase the time of contact of runoff and encourage infiltration, deposition, and biological uptake of nutrients. In general, vegetative controls can achieve moderate removal rates of sediment but they may not be very effective for treating nutrients (EPA 1989).

Grassed swales can treat storm water runoff before it enters a structural device. Studies on grassed swales do not provide consistent results to estimate nutrient removal efficiency. Moderate removal of particulate nutrients can be expected if grassed swales are built with low slopes, erosion resistant grass species, highly permeable soil and check dams (Schueler 1987).

Buffer strips are designed to accept and treat runoff in overland sheet flow. Overland sheet flow can be enhanced by building a level spreader (e.g. stone trench or low gravel berm) to distribute runoff entering the buffer. Walker (1990) estimates total Kjeldahl nitrogen removal of approximately 30-60% and total phosphorus removal of 36-67% based on simulations with the model P8 for a case study in Rhode Island. Given that the design and maintenance of buffer strips dramatically effect their performance, conservative estimates would be at the lower end the reported effectiveness ratings.

Natural or created wetlands may be used to treat runoff. Schueler (1987) provides references for design specifications and planting requirements for wetlands and shallow marshes.

Infiltration Devices. Infiltration (or retention) devices include infiltration basins, trenches and porous pavement. The physical process of infiltration through porous media removes nutrients and controls storm runoff depending on the design and maintenance. If storm water is treated in a basin or other structure to remove most of the large sand and colloidal particles additional sediment-attached phosphorus treatment can occur as water percolates downward through soil (EPA 1989). Up to 90% of the dissolved nutrient fraction may be removed by infiltration as evidenced by the relatively low concentration of nutrients in ground water below infiltration devices (Walker 1990). However, some of the nutrients in percolate may reach surface water. Extensive pretreatment is required to remove coarse-grained sediment before storm water enters an infiltration device to prevent clogging and device failure (Schueler 1987; EPA 1989).

Infiltration basins are typically large natural or excavated depressions designed to remove dissolved and fine-grained sediment-attached nutrients from areas up to 20 ha. The basin holds stormwater as it percolates through the bottom or sides of the basin. Detention basins can be designed to control peak discharges based on the volume of a large design storm and for maximum storage (Schueler 1987). Basins effectiveness reach 60-75% of influent total nitrogen and total phosphorus removed for a 2-year design storm (EPA 1989; Walker, 1990). Siting considerations include soils with sufficient hydraulic conductivity and depth to ground water and bedrock. Detention basins have a high rate of failure compared to other urban BMPs (Schueler 1987) due to clogging with coarse-grained sediment.

Infiltration trenches and dry wells can treat nutrients from areas of up to 4 ha. Trenches have similar site restrictions as basins and both are prone to clogging. When designed to treat 1.3 cm (one half inch) of stormwater per impervious acre, infiltration trenches can remove 45-55% of influent total nitrogen and phosphorus. EPA (1989) found greater efficiencies of up to 70% have been reported for treating a 2.5 cm storm.

Porous pavement treatment effectiveness is not known with confidence. Siting of porous pavement is limited to gentle slopes, permeable soils and light traffic and may require street sweeping or special maintenance to help prevent surface sealing.

Detention Devices. Storm water detention serves as temporary storage or active treatment. Storage treatment effectiveness is limited and device types include an inlet control at the source, on-site detention for several parcels of real estate, within the sewer or conveyance network, or off-line storage when a predetermined flow has been exceeded at a treatment facility.

Dry basins do not retain a permanent pool of water and their primary pollutant removal mechanism is sedimentation, which limits their effectiveness compared to other devices. Stahre and Urbonus (1990) estimate total nitrogen and phosphorus nutrient removal efficiency of 10-20%. They suggest the use of the lower end of the efficiency rating for planning purposes. EPA (1989) provides estimates of greater efficiency (30%, up to 70%) based on a ratio of basin volume to basin runoff, however they caution that their estimates are probably high.

Extended detention basins temporarily hold water for an extended period of time which allows further sedimentation. Normally, extended detention basins are dry and they can be effective for removing particulate nutrients and for reducing peak discharges (Brach 1989). Pollutant removal efficiency increases with detention time. Total phosphorus removal at 12 hours of retention is approximately 40% and increases linearly to about 55% efficiency at 48 hours. Total nitrogen removal is about 22% removal at 12 hours, increasing to 32 at 24 hours and increasing linearly to 40 at 48 hours OWML (1983).

Wet basins (ponds) permanently hold a pool of water and their effectiveness is rated relatively high compared to other urban BMPs. Walker (1990) reports typical wet detention pond treatment efficiency for total phosphorus ranging between 36%-70% and 31-47% for total nitrogen based on simulations of the model P8 and assuming a treatment efficiency for total suspended solids of 70% and 85%.

The effects of macrophytes on the removal efficiency of detention ponds has been studied in Australia. Phillips and Goyen (1987) and Lawrence (1986) found an increase in treatment efficiency of approximately 5-30% for ponds with macrophytes compared to ponds without macrophytes having the same hydraulic detention time.

Forestry. Forest land management, practice 409, is a comprehensive RMS that includes practices for erosion control and production. Forestry practices should be evaluated individually and in combination to determine their effect on surface and ground water quality (SCS, 1988) and nutrient runoff.

Forestry BMPs reduce nutrient losses, minimize overland flow and the dislodgement and transportation of soil, and stabilize disturbed areas. Site slope and aspect, and the potential for erosion should be carefully evaluated to estimate the impact of forest. Silvicultural activities that cause disturbance include building roads and harvesting trails, tree removal, site preparation and reforestation, revegetating disturbed areas, and wildfire protection.

Very few studies on the impact of road building on sediment loss are available. Impacts are expected to be very site-specific and the potential for extensive sediment losses is high. Even less is known about the effect of road building nutrient losses.

Most work on forestry BMPs report that a combination of practices minimized adverse impacts. Likens et al. (1970) and Pierce et al. (1970) were the first to document an increase in stream nitrate concentration after clearcutting in the White Mountains of New Hampshire. Continued work in the White Mountains showed considerable increases in stream nitrate after the first three year of clearcutting (Martin and Pierce 1980). In other parts of the country, the pattern of increases in stream nitrate concentration after clearcutting were not as pronounced (Martin et al. 1984).

Stream nitrate concentrations averaged 2 mg/l during the second and third years after clearcutting in 15 New England watersheds. Typically loggers used streams as a cutting boundary and left a buffer zone if tracts on both sides of the stream were to be logged. For most of the harvests, <50% of the watershed area was clearcut. Runoff nitrogen from clearcut watersheds averaged < 2 and generally < 0.4 mg/l. Reference stream concentrations averaged 0.5 mg/l nitrate nitrogen. The highest values found were 6.1 mg/l in the White Mountains (Martin et a1. 1984).

In a 15-year study in Pennsylvania, Lynch and Corbett (1990) evaluated the effect of using a stream side management zone (SMZ) of 100 feet on both sides of a perennial stream. In the SMZ, trees were harvested that could threaten the stream channel flows if they fell. Equipment movement over streams was prohibited unless by bridge or culvert. Trails, roads and log landings were planned by a professional forester and harvesting was prohibited when soils were excessively wet. Lynch and Corbett (1990) found mean annual nitrate concentrations from 0.28-0.4 mg/l were significantly higher than the background range of 0.05-0.11 mg/l for the first two years of harvest. Nitrate was also higher than background for the next 9 years but ranged from 0.8 to 0.14 mg/l.

In a review of harvest and site preparation BMPs used in the South, Reikerk et al. (1989) found that mountainous region runoff and nutrient losses were substantially elevated after harvest. Total nitrogen runoff concentrations ranged from 0.5-1.0 mg/l and approached baseline levels of approximately 0.2 mg/l three years after harvest. Nitrogen export ranged from 0.3-1.4 kg/ha/yr and returned to baseline levels of around 0.15 kg/ha/yr . Phosphorus losses were not affected appreciably in the mountains and fluctuated around .01 mg/l (1.3-1.7 kg/ha/yr). Piedmont region nutrient export was generally unchanged by forest practices. Shearing and windrowing increased nitrogen export to near 0.2 mg/l (.3 kg/ha/yr) in the first year after harvest from a background near zero in the upper coastal plain studies in Texas. For these sites dissolved phosphorus increased from .01 to .03 mg/l after the first year of windrowing. Nitrogen and phosphorus export in the lower coastal plain are relatively insensitive to the disturbance of intense site preparation (Reikerk et al. 1989).

Glasser (1989) also reviewed literature on the effect of BMPs on water quality for major land resource areas in the southern US and found most of the increases in stream nitrate concentration due to forest practices were in the mountainous regions .

McClurkin et al. 1983 monitored the effect of clearcutting on sediment and nutrient for four small 0.12-0.6 ha (0.3-1.5 acre) watersheds near Lexington, TN. BMPs included tree length skidding on the contour, no stream crossings, and residue and the forest floor was left intact while planting was done by hand. One third of the TKN and more than two thirds of the total phosphorus losses were via the sediment-attached phase. Dissolved TKN and phosphorus losses were similar to the untreated control.

Miller et al. (1988) studied sediment export from a clearcut, selection cut and an uncut control watershed in the Ouachita Mountains of Arkansas for three years following harvest. For the first year after harvest, clearcutting significantly increased sediment export, but not in the following two years. Clearcut sediment losses averaged 237 kg/ha/year for the first year after harvest. Stormflow suspended solids were less than 100, 50, and 20 mg/l for nearly all events.

Shearing and windrowing clearcut debris resulted in significantly higher mean annual sediment losses compared to a clearcut watershed treated with chemicals for site preparation or control watersheds on the southwestern plateau of Arkansas. (Beasley et al. 1986). On the flat Gulf Coastal Plain of Arkansas. Beasley and Granillo (1988) found clearcutting and mechanical site preparation significant increased runoff and sediment yield and average sediment losses were 264 and 64 kg/ha for the first and second post harvest years. Runoff and sediment yield from selective harvest were not significantly different from the undisturbed control .

In north central Washington a study of the effects of helicopter and longspan skyline harvesting along with a riparian buffer the width of the stream valley evaluated the effects of these combined practices in an area where erosion harvest was moderate to high (Fowler et al. 1988). For a period of three years after the harvest, total nitrogen loading was not significantly different in the treated watersheds compared to the controls.

Moore (1975) compiled the results 22 monitoring studies in Western Washington and Oregon, two in eastern Washington, three in Northern Idaho and two in Alaska to summarize the findings of fertilization studies its affect on stream chemistry. Typically, fertilizer nitrogen was applied by helicopter at a rate of 168 to 224 kg/ha. Pretreatment total nitrogen concentrations for a representative watershed were 0.01 mg/l. Peak nitrate in streams usually occurred 2 to 4 days after application. Concentrations decrease, but remain above background for about 6 to 8 weeks. Maximum peak nitrate concentrations were approximately 1 mg/l with the highest peak level at 4.0 mg/l. Buffer strips that were not fertilized were effective in keeping losses to less than 0.5%.

Streamside management zones can be an important part of a system to reduce nutrient losses and provide other water quality benefits (e.g. streambank stabilization, wildlife habitat preservation) . Management objectives determine SMZ design, however trapping sediment and sediment-attached pollutants are likely for NPS control. Primary factors for SMZ design are width (length of flow), extent along water courses and BMPs within the SMZ.

Streamside management zone width is based on slope and soil erodability, and as each of these increase, proper design width generally increases. Nutter and Gaskin (1989) suggest that soil infiltration capacity will greatly affect nutrient trapping. Fine soil particles will also travel further, making soil textural properties important for design (Hewlett 1982).

Since most forest erosion occurs through channelized flow and not sheet flow (Brown, 1980) the SMZ should protect perennial, intermittent, and ephemeral streams where forest activities increase upland water runoff. Harvesting can increase water runoff and cause intermittent streams to become perennial (Lynch et al. 1985).

Partial harvesting within the SMZ is recommended to remove on-site nutrients, encourage stand growth, and increase nutrient uptake. Omernik (1981) and Fail (1987) found nutrient uptake in a mature stand is significantly less than one in earlier stages of growth.

Lanier (1990) reports that published literature on SMZ effectiveness for trapping nutrients is lacking. Recommendations for SMZ design are based on experience and very little monitoring. Therefore, careful management to reduce NPS impacts will employ SMZs in a system with silvicultural source controls rather than counting on the SMZ trap pollutants and protect the water course.

Agriculture. The lessons learned from the Model Implementation Program (MIP), the Great Lakes 108a projects, and the Rural Clean Water Program (RCWP) provide a strong basis for improving agricultural NPS control methods on a watershed scale.
MIP projects found they should identify and treat critical areas that directly contribute to water quality results. Five of the MIP projects demonstrated from field and plot studies that BMPs could provide water quality benefits (NWQEP 1989).

The 108a projects found that conservation tillage was effective at reducing erosion and total nutrient concentrations however it may increase dissolved nitrogen and phosphorus concentrations in surface waters, demonstrating the need for nutrient management (Newell et al. 1986). Conservation tillage and fertilizer management were the most cost effective methods for cropland phosphorus control. Projects should address the differences in pollution sources to increase the chances of meeting goals.

Some of the lessons learned from RCWP include: a) a reduction in erosion may or may not improve water quality, b) the effectiveness of structural BMPs (e.g. waste storage structures, sediment basins, terraces, improved irrigation) may be enhanced by nutrient management, waste management and conservation tillage c) practices that reduce surface runoff can increase the leaching potential for nitrate and pesticides .

The National Water Quality Evaluation Project (1989) found that the level of participation by farmers in a voluntary program is important for success. There is no single accepted relationship between treatment strength (number of farmers participating) and pollutant reduction. Treatment of 75% of the critical area was sufficient for some RCWP projects and not for others. Nonpoint source control was not demonstrated when: a) a water use impairment was not documented, b) the project area was too large and contained varied types of pollution sources, c) where local interest was inadequate, and 4) where monitoring capability was unable to detect trends.

Conservation Tillage. Conservation tillage, practice 329, is any tillage and planting system that leaves at least 30% of the soil surface covered by residue after planting (SCS 1987). The soil is tilled only to the extent needed to prepare a seedbed, incorporate chemicals, control weeds, and plant the crop (Brach 1989). The purpose is to reduce soil erosion, help maintain good soil tilth and efficient moisture use, improve water quality, and provide food and cover for wildlife. In general, conservation tillage increases infiltration and decreases evapotranspiration resulting in reduced surface runoff but can potentially increase percolation.

The average reduction in surface runoff due to conservation tillage may be on the order of 20-25% (Baker and Laflen 1983; Wendt and Burwell 1985; Gilliam and Hoyt 1987).

Erosion control and the reduction of edge-of-field sediment loss is the primary attribute of conservation tillage compared to conventional tillage with the moldboard plow. For every 9 to 16% increase in residue cover, erosion is approximately halved (Baker and Laflen 1983). Conservation tillage should therefore reduce erosion by 75 to 90% depending on the percentage residue cover over that for conventional tillage (Dillaha et al. 1988).

Sediment-attached nutrient losses are generally reduced as more sediment remains in place. With conservation tillage, it may be difficult to incorporate nutrients and maintain plant residue, and the quantity of nutrients near the soil surface may increase. If the surface soil has a higher concentration of nutrients then the nutrient content of eroded sediment is likely to be higher. Losses of sediment-attached nutrients due to conservation tillage will be affected by the soil lost and its nutrient content (Baker and Laflen 1983).

The nutrient content of surface runoff is also likely to be higher if fertilizer is not incorporated. Runoff nutrient concentrations are directly proportional to soil nutrient concentrations (Baker and Laflen 1983). For conservation tillage fertilized with manure, losses of nutrients in surface runoff are greatly reduced with little incorporation (Walter et at. 1987). Soluble nutrients leaching from crop residue may also increase soil fertility (Barisas et al. 1978; Smith et al. 1974). Losses of dissolved nutrients are a function to the total runoff volume. Unless increased nutrient concentrations in conservation tillage are not offset be decreases in runoff volume, dissolved nutrients flux will increase. Conservation tillage and nutrient management combined should reduce dissolved nutrient losses to a level approaching conventional tillage.

Terraces. Three types of terraces, practice 600, are considered for their effects on nutrient losses: gradient with full flow outlets, and level storage with or without an underground outlet (SCS 1988). The primary mechanism for the control of nutrients is deposition of sediment-attached nutrients. Terraces can have a detrimental effect if they convey runoff directly to surface waters or if nitrate leaching becomes a problem.

Midrange decreases in surface runoff are 20 to 50% for terraces with underground outlets. Increases in deep percolation are more variable and can range from 5 to well over several hundred percent (SCS 1988).

Although terraces require extensive soil disturbance and are expensive to build, they reduce transport of eroded soil and sediment-attached nutrients from fields. Gradient terraces increase infiltration and percolation and decrease water runoff and they transport greater amounts of sediment-attached nutrients compared to storage terraces. Level terraces can trap up to 95% of sediment-attached nutrients and from 30 to 90% of dissolved nutrients (SCS 1988).

Decreases in dissolved nitrogen concentrations from terraces compared to corn grown on a contour were 14-50% and sediment-attached nitrogen concentration losses were decreased 7-13% (Baker 1985). Level terraces can reduce total nitrogen loadings by 85% (NWQEP 1982b) . Terraces are recommended as nitrogen controls where no potential ground water problems exist.

Total phosphorus losses can be reduced 67% by terraces as compared to contour farming (NWQEP 1982b) . Dissolved phosphorus losses can decrease by about one-third on a concentration basis (Baker, 1985) and from 40-60% on a mass basis (SCS 1985). Sediment-attached phosphorus concentration losses were reduced up to one-third (Baker 1985) and nearly 100% (SCS 1988).

Irrigation System Furrow Improvements. Nutrient losses from irrigation return flows continue to be a serious problem. BMPs to reduce furrow erosion also reduce export of sediment-attached nutrients. Unless meteorological factors dominate, farmers have greater control of system hydrology (e.g. irrigation scheduling and rates) and return flow quality compared to farmers in nonirrigated humid areas.

To improve the quality of water draining irrigated farmland, a system of improvements may be developed to treat site-specific problems. The system includes the crop, tillage, and chemical application, irrigation water management, and improving, reorganizing, or converting the existing furrow irrigation system.

The Rock Creek RCWP implemented improved irrigation, sediment basins, conservation tillage and filter strips and reduced suspended sediment concentrations in return flows by 75% (EPA 1990) and mass loadings of total phosphorus to the Snake River by 57% (Maret 1990).

Reducing the number of irrigation events, irrigating every other furrow and additional water management BMPs have also been proposed. The applicability depends on the crop and water requirements. Models to estimate irrigation runoff and nutrient losses in the irrigated West have not been developed to the extent of models in the humid nonirrigated areas .

Water runoff from irrigated land has been estimated as 30-40% for the Magic Valley of Idaho depending on the crop (McNeal 1982). Brockway (personal communication 1990) estimates return flows at field ends for furrow gated pipe, and concrete ditch systems are 35-45% of the applied water. He suggests that water reuse can reduce losses to 30% and cutback or surge irrigation return flows can be reduced to 15% or less.

Control of slope through land leveling, grading, or contour furrowing can reduce erosion losses. Israelson et al. (1946) showed that erosion losses increased with increasing furrow slope, and an increase in slope from 1 to 6% increased erosion 16 times. From a compilation of several studies, McNeal et al. (1982) found erosion to vary by the 2.2 power of the slope. Kemper et al. (1985) found that erosion was commonly about a 2 to 3 power function of furrow slope.

Shortening the field or furrow run length also reduces erosion losses. Using a multiset system of gated pipe, that can be removed to avoid damage by farm equipment, can reduce erosion by 40% (Brockway 1986). Buried pipe can also convey water, reduce erosion losses, and farm equipment can cross without damaging it. An approximate 83% sediment reduction efficiency with a range of 74-83% was estimated for buried pipe in the Idaho RCWP by Carter and Berg (1982).

Crop and straw residues reduce the erosive energy of advancing irrigation water. Berg (1984) found that straw-treated furrows reduced erosion by 30-70% compared to untreated furrows. Straw significantly increased infiltration and lateral water movement. Straw-treated furrows also reduced net sediment yields 52 to 71% with the greater soil savings during higher irrigation flow rates (Brown 1985). Sediment concentrations decreased 50 to 33% at low and high flows respectively. Miller et al. (1987) found that straw residue was effective at reducing sediment yields up to 100 fold compared to untreated furrows.

Sod furrows were shown by Cary (1986) to nearly eliminate erosion, but mowing was required. Berg and Carter (1980) found that seeding crops in furrows was effective in reducing erosion.

Much of the sediment lost from irrigated fields occurs in the last few meters of the furrow (Carter and Berg 1983). Tailwater ditches should be kept shallow to avoid a severe gradient and associated increased scour and erosion. Tailwater ditches are often kept well cleaned and below the level of the end of the furrow to remove tailwater rapidly, but this practice erodes furrow ends and creates convex-shaped fields. To remedy the problem and to increase the productive area of the field, Carter and Berg (1983) developed a buried pipe system with riser inlets to drain fields and trap up to 80 to 95% of the sediment in runoff.

Tailwater pollutant control devices include collection ditches, buffers or sediment traps. Efficiency is high for these structural BMPs but they are subject to failure and require maintenance. Concrete collection basins may reduce sedimentation by 25% (Brockway 1986). Mini-basins and I-slots can trap up to 86% of the sediment from irrigated furrows. Efficiency for sediment basins has been reported as 66% with a range of 53 to 85% by King et al. (1984). Carter and Berg (1983) report a 87% efficiency with a range of 75-90% sediment trapped. Sediment traps require cleaning to maintain high levels of efficiency. King et al. (1984) found 51% phosphorus trapping efficiency for sediment basins. The lower trapping efficiency compared to sediment is thought to be due to was lower settling velocity and subsequent flow-through of phosphorus attached to clay sized particles .

Nitrogen content of irrigation water can have an affect on dissolved nitrogen concentration in surface irrigation return flows. Carter et al. (1971) found that nitrogen concentration did not increase appreciably as water passed over the soil surface. Fitzsimmons et al. (1972), Carlile (1972), and Naylor and Busch (1973), also found little difference between dissolved nitrogen in irrigation and return flow waters. However, when soluble nitrogen is added to irrigation water or when liquid nitrogen is applied, increases in return flow concentrations can be expected (Carter and Bondurant 1976) and runoff losses may be proportional to the amount applied (Naylor et al. 1972).

Nitrogen concentrations may be higher when irrigation water makes contact with decaying plant material. Higher organic nitrogen losses may be attributable to runoff losses of organic matter and sediment (Carter and Bondurant 1976).

Water management practices have a significant impact on runoff, nutrient losses and farm productivity. Numerous power functions have been fit to the relation between runoff water applied (stream size) to erosion. McNeal et al. (1982) found that runoff volume raised to the second power gave a reasonable estimate of erosion. GMC Neal et al. (1982) summarize the combined effects of slope and runoff water applied on sediment loss.

Animal Waste Management Systems. Waste management systems, practice 312, serve to collect, control, and treat waste from confined animal lots such as feedlots and barnyards. The purpose is to minimize pollutant runoff and to conserve nutrients for crop production. Feedlot and barnyard runoff practice standards are developed to provide planning tools to prevent the discharge of animal waste directly into waterways.

Diverting tributary or upland runoff water away from concentrated animal areas reduces nutrient runoff and can improve conditions for waste handling. Dikes, diversions, earthen berms, ditches, or inlets and drains may be used to divert tributary flows. Relocating a barnyard away from a watercourse may be necessary to meet water quality goals. Fencing may be needed to exclude cattle from a watercourse except when a stream crossing is necessary.

Livestock should be kept in a designated area to facilitate the collection of manure and to protect water diversion structures. A scraping schedule should be determined for removal of lot manure. Curbs may also be necessary to control manure and divert it to collection locations. Contaminated runoff and seepage should be stored for treatment, disposal, or use on cropland.

Robillard et al. (1983) found the diversion of upland flows, the collection and diversion of roof water and the interception and diversion of subsurface flows from two barnyards reduced total phosphorus export by 80%. BMPs reduced transport of nutrients by reducing flows while nutrient concentration remained essentially unaffected.

Upland diversions and roofing reduce manure contact with precipitation and decrease transport. Young et al. (1982) determined that overland flow from upland, tributary areas, passing through the feedlot can become sufficiently mixed within and approach the same pollutant concentration.

Treatment for feedlot, barnyard or milkhouse waste not applied to cropland must be carefully designed in consideration of hydraulic loading. One system for treating animal lot runoff is a settling basin to trap solids and dissolved pollutants followed by vegetative treatment (e.g. filter strip, grassed waterway) to further treat effluent. Settling basins reduced ammonia nitrogen concentration by 40%. (Miner et al. 1981) and total nitrogen and total phosphorus by 35-40% (Edwards et al. 1983).

Filter strip treatment of barnyard and feedlot waste can be quite variable. Meals (1987) found significant reductions in nutrient concentrations only during the growing season. Only annual subsurface output of total phosphorus was significantly different from inflow concentrations. Total phosphorus mass reductions were 12% and total nitrogen reductions were 15%. Meals (1987) reports high hydraulic loading rates may have been part of the reason for poor performance. Willrich and Boda (1976) and Walter (1983) also report decreased treatment efficiency at high loading rates. Channelized flow decreases contact time and performance.

For a grass filter strip Doyle et al. (1977) found phosphorus reduction at 62%. Willrich and Boda found 62% of phosphorus removal with loading rates 5-15 cm/week and 46% phosphorus removal at loading rates of 20-25 cm/week. Other studies report higher filter strip treatment effectiveness in the range of 75-80 for sediment and nutrient concentrations (Yang et al. 1980; Dickey and Vanderholm 1981; Edwards et al. 1983; Dillaha 1986). Still others Overcash et a. (1976) and Young et al. (1978) report effectiveness for reducing total phosphorus at 87 and 88%. For studies using forest buffers and orchard grass, corn stubble and tilled plots, Dolye et al. (1977) and Thompson et al. (1979) found soluble and total phosphorus levels were reduced to background levels.

For treating milkhouse waste, Barker and Young (1985) showed a settling basin in line before a filter strip removed total phosphorus up to 68-98% by concentration and 98% by mass. Total nitrogen effectiveness was up to 45-98% by concentration and 99% by mass. Schwer and Clausen (1989) also treated milkhouse waste with a vegetative filter. Phosphorus removal on concentration basis was 86% and 89% by mass. Total nitrogen removal was 83% on a concentration basis and 92% by mass. For both milkhouse waste filter studies authors stress the importance of keeping within the hydraulic loading capacity of the filter.

Nutrient Management Systems. Nutrient management systems address requirements for soil fertility and protection of water quality. Nutrient sources include organic wastes, chemical fertilizer, soil reserves, and crop residues. Nutrient management systems reduce the availability and movement of nutrients by controlling the amount, source, form, placement, and timing of application to crop and forage areas.

The rate of nitrogen application should match the needs on the crop based on realistic yield goals. Losses of both nitrogen and phosphorus increase with increasing application beyond what is used by the plant (NWQEP 1982b; Baker and Johnson 1983, WDATCP 1989). For the application of both manure and chemical fertilizers, soil testing is an essential BMP to assess crop nitrogen and phosphorus needs. Credit for nitrogen input by legumes and manure should be considered in the nitrogen recommendation. Proper rates can reduce nitrogen losses by 35-94% as compared to excessive rates (NWQEP 1982b) . Excessive rates of application increase the potential for nitrate leaching into ground water supplies.

Losses of nutrients are also highly influenced by the timing of application in relation to runoff events. As the time between nutrient application and runoff increases, especially if there is crop uptake, the chances of losses are decreased significantly. The timing of application should be just prior to or during the maximum nutrient uptake, as either spring or summer. Split nitrogen application can reduce potential losses by up to 30% as compared to a single application (NWQEP 1982b) .

The National Water Quality Evaluation Project (1982a) found when applying manure in the fall, up to 50% of the total nitrogen can be lost through decomposition and leaching. Winter manure applications have also caused large losses up to 86% of the nitrogen and 94% of the phosphorus can be lost in a single runoff event.

The method of fertilizer and manure application greatly affects the potential for losses in runoff. Broadcast fertilizer should be incorporated wherever possible. Timmons et al. (1973) found disk incorporation of fertilizer decreased dissolved nutrient losses 25-50% compared to surface application and plow-down incorporation decreased losses by about one half.

If manure is broadcast it should be incorporated immediately. Point injection of liquid fertilizer at 5 cm depth did not decrease nutrient losses compared to unfertilized plots (Baker and Laflen, 1982). However research shows that the incorporation or injection of animal wastes can eliminate losses of nitrogen through erosion and volatilization while increasing crop yields (NWQEP 1982a) .

Watershed demonstrations show that application rates can be reduced in a voluntary program. Farmers in the Delaware RCWP have used soil and manure testing to determine manure and fertilizer application rates. Split application of nitrogen with side banding has minimized losses by matching fertilizer placement and timing to crop needs. Phosphorus applications have been reduced by 50%. After a seven year period BMPs have not decreased total nitrogen to the watershed outlet. Total phosphorus has decreased by 71% (Ritter et al. 1989) but the changes in total phosphorus may be attributable practices associated with the reduction of suspended sediments, but also a reduction in fertilizer application (WRANCC 1986).

In the Minnesota RCWP, the development of nitrogen budgets for farmer's fields includes accounting for manure and legumes. Early spring application of nitrogen dropped by 50% and total nitrogen applied decreased by 20% (NWQEP 1989). Other major practices in the watershed include conservation tillage, contour strips, permanent vegetative cover, and animal waste management systems. Thus far, only annual median daily loadings of suspended solids and nitrite have shown a significant decreasing trend. AGNPS modeling comparing pre and post-implementation conditions for a 2-year, 24 hour storm estimate an 8% improvement in dissolved nitrogen concentration and a 13% improvement in total nitrogen in sediment loading. Soluble phosphorus concentrations improved 22% and the total phosphorus in sediment improved 15% (Wall et al. 1989).

The soil acts as a buffer or a large nutrient reservoir, releasing nitrate-nitrogen to the ground water on a continual basis and where phosphorus has been over-applied, immediate loading reductions due to nutrient management cannot be expected (NWQEP 1989).

To develop a strong nutrient management program, utilize the most advanced technology accessible and provide a framework for reporting progress. Changes in management practices can conflict with traditional farming methods. Sufficient time is required to educate farmers about efficient methods to manage farm costs and protect water resources .

Briefly, the components of a nutrient management program are (NWQEP 1988):

a. Determine current nutrient management practices.

b. Determine the range of soil fertility and crop nutrient requirements .

c. Identify nutrient sources and utilization areas for potential redistribution from areas of oversupply areas of undersupply. Determine how much improvement in nutrient management can be accomplished given current practices in the watershed.

d. Provide cost-sharing and technical assistance for cost-sharing and technical assistance for soil testing and manure testing .

e. Estimate nutrient mass balance fore each filed, farm, and for the entire project area. Example dairy or mixed livestock components are:

Inputs: feed, fertilizer
Outputs: milk, meat, crops, excess manure
Recycle: manure, crop residue

f. Determine the current inputs for the area generating pollutants, or for each field in the case of a small project area. Compare with required nutrient needs of crops and pasture based on the soil test results, soil type and land form. Adjust nutrient applications accordingly.

g. Use worksheets and written plans to formalize records on nutrient input requirements and application rates for individual farm operations or fields. Where possible the plans should identify the range of susceptibility of ground and surface water to nutrient over-application based on soil type and land form. Beegle at al. (1989), Schulte et. al. (1989), Harmon et al. (1988), and Vitosh and Carroll (1987) developed nutrient management worksheets using spreadsheet software that are available to make the calculations and planning less cumbersome for managing farm fertilizer and manure applications .

h. Educational materials, field demonstrations, and technical assistance along with the calibration of manure and fertilizer spreaders is important. Demonstrations may be needed to show the nutrient value of animal waste, advanced fertilizer applications such as banding, split application, injection or improved incorporation the use of legume cover.

i. Promote the effective use of animal waste storage facilities for the optimal timing of manure nutrient utilization. Also, promote tissue testing to determine the proper nutrient requirement and to verify nutrient applications .

Grassed Waterways. Grassed waterways, practice 412, are used primarily to prevent rill and gully formation, to stabilize an active gully, as a stable outlet, or to convey runoff from terraces, diversions, or other outlets without causing erosion or flooding and to improve water quality. Grassed waterways must used in conjunction with upland soil conservation practices (Baker and Johnson 1985).

A secondary function may be to trap sediment and sediment-attached nutrients in runoff. Depending upon the rate of deposition, the capacity of the grassed waterway may be exceeded, and can become buried and rendered ineffective (Lake et al. 1977). The Black Creek, Indiana watershed study found that grassed waterways were more effective in reducing sedimentation if used with other BMPs such conservation tillage, diversions and terraces. DelVecchio and Knisel (1982) found minimal treatment by a grassed waterway compared to contouring and crop residue management. They found little effect on dissolved nutrient concentrations due to grassed waterways. Davenport et al. (1984) simulated the effect of grassed waterways with several tillage scenarios using the model CREAMS. They found that where overland flow sediment yields were high as in the case of conventional tillage systems, grassed waterways could remove approximately half of the overland flow sediment yields. However, Davenport (1984) found that high rates of deposition in grassed waterways could cause long-term maintenance problems. Hamlet et al. 1987 report lower trapping effectiveness during extreme rainfall events and resuspension of previously deposited sediment from a grassed waterway.

Most studies using grassed waterways report the effectiveness of a system of BMPs that include a grassed waterway. Since flow is channelized, only minimal long-term sediment-attached nutrient removal can be expected from grassed waterways. Effectiveness is a function of maintaining channel shape, vegetation and upland soil erosion control. Asmussen et al. (1977) report 2-6% effectiveness for removing sediment from a plot study. Using CREAMS to simulate effectiveness Davenport (1984) found a 1% sediment removal and a similar removal rate for nutrients for a grassed waterway below a no-till system.

Filter Strips. Filter strips are designed to remove runoff-borne nutrients and sediment in combination with other source area BMPs. Interest in the use of filter strips has been increasing due to their relative low cost and their effectiveness in removing pollutants in carefully controlled short-term research studies. However, there is a great deal of uncertainty about the long-term (10-20 years) effectiveness of most filter strips now in use. Design is an important concern. The minimum standards of VFS design are provided by the SCS (1987). This section focuses on cropland filter strips referred to as vegetative filter strips (VFS) .

To estimate the effectiveness of VFS each site must be inspected before installation and periodically throughout its life to evaluate functional attributes. Dillaha (1989) identifies the processes in flow hydraulics thought to control pollutant removal in VFS. Pollutant removal is likely a function or infiltration of runoff and pollutants, deposition of suspended sediments and sediment-attached nutrients, filtration of suspended pollutants by vegetation, adsorption on soil and plant surfaces, and absorption or uptake of nutrients by plants. The basic design criteria to improve the performance of VFS are slow uniform overland flow over a sufficient area and sufficient contact time.

Bingham et al. (1980) found reductions in nutrient concentrations from a grassed buffer were attributable to dilution in the buffer area in addition to functional mechanisms .

Dillaha et al. (1985) showed nutrient concentrations leaving the filter strip exceeded the nutrient concentrations leaving the strip. They attributed the increase to the flushing of accumulated nutrients or to nutrients leached from vegetative cover.

Lanier (1990) found filter length, uniformity of flow in the field and along the filter length, slope of field, type and density of vegetation, sediment size distribution and the amount of dilution runoff from the contributing area are primary variables affecting VFS effectiveness. Source area management and meteorological factors that affect influent pollutants, also determine VFS performance. The cropland source area and the VFS must be viewed together, the entire system is interactive (Lanier 1990).

Management of the source area is key to filter strip effectiveness. Where cropland hydrologic condition is poor, and erosion rates are high, long term performance of the VFS would be expected to be low. Dillaha et al. (1989) found that sediment deposition occurred upslope or in the first few meters of the VFS. After the vegetation in the upper part of the VFS was buried, deposition advanced down slope reaching the end of the VFS (Dillaha et al. 1989). Niebling and Alberts (1979) and Tollner et al. (1977) report similar findings .

Dillaha (1989) found that where surface runoff accumulates in natural drainageways channelized flow results and the VFS can be become ineffective. Topographic factors may limit VFS performance and channelized flow may be more of a problem in hilly areas compared to flat area (e.g. coastal plain) .

Where shallow overland flow can be achieved, the effectiveness of VFS may be estimated in a relatively straightforward manner by considering sedimentation and filtration as the primary removal mechanisms. The link to nutrient removal is based on nutrient losses closely associated with sediment. Soil particle size distribution affects removal and nutrient removal as well. Walter et al. (1979) found that soils with the highest adsorption capacity have small particles with low density. Most of the pollutant associated with sediment will be transported on smaller, dense particles (Dean 1983).

Flanagan et al. (1989) presented a simplified model to estimate VFS effectiveness for removing sediment. Sediment-attached nutrient removals are expected to closely match sediment removal efficiencies. The methods to estimate effectiveness are based on the CREAMS and its equations that consider influent particle size distribution, strip width, and slope length. Also for a desired trapping efficiency, the method determines VFS width.

For sediment of nonuniform particle size, the sediment delivery ratio (SD) is:

$$SD = \sum_{i=1}^{5} f_i x_u \Phi i$$

where:

f_i = the fraction of particle size i entering the strip and is the particle index (dimensionless)

x_u* = source area length of flow divided by total length (total length = source area length of flow + VFS length of flow) (dimensionless)

ϕi = measure of depositability for each particle size i (dimensionless) is given in Table 4.2.6

The quantity f_i is determined by soil type and the slope of the source area and VFS (these slopes are assumed to be the same) . For a silt loam soil the computed fraction of primary clay, primary silt, and small aggregates is given in Table 4.2.4.

Flanagan et al. (1989) provides example SD calculations for a field with silt loam soil in a moderate slope of 2.5% with slope length of 100 m. For the example, the desired VFS effectiveness is a SD of 0.5 and reduce sediment losses from 10 t/ha to 5 t/ha. The particle size fractions reaching the strip are:

$$F_{clay} = 0.05$$

$$F+_{silt} = 0.24$$

$$F_{sm.agg} = 0.36$$

If the VFS is 2 meters wide then x_u* = 0.98 and o the values for primary clay, primary silt and small aggregates are 0.11, 2.84, and 13.5, the SD would be estimates as 0.55 as in the sample calculations Table 4.2.6.

Dillaha (1989) proposes that additional site selection, installation, and maintenance, and criteria be incorporated into VFS implementation.

Site Selection

- Soil drainage and depth to water table must be sufficient to allow adequate vegetative growth and prevent extended periods of saturated soil

- VFS site should be devoid of hillside seeps of other continuous discharges

- Consider the reduced effectiveness of snow and frozen ground

- Consider slope (e.g. is there enough distance available to accommodate design length based on slope, is slope too severe)

- Consider if VFS will work properly for large fields with natural drainageways and grassed waterways

- VFS should not be installed above a source area they are designed to protect

Installation

- VFS should be installed on the contour of the lowest elevation of the source area to prevent concentrated flow

- Some grading may be necessary to alter natural drainages both above and below the VFS

- A level spreader formed of material resistant to erosion can be used to disperse concentrated flow into more shallow overland flow

- Berms or diversions may be needed at 15 or 30m intervals to intercept and divert flow over the VFS where flow along or parallel to the VFS is significant

Maintenance

- Vegetation should be mowed and removed from the site to decrease nutrient export form the VFS outlet

- Carefully controlled grazing of the VFS may be considered to remove vegetation, especially if conditions and effects are known

- The effects of herbicides on adjacent fields or the VFS should be considered

- Herbicide sprayers should be turned off when crossing the VFS or source areas

- Changes in flow patterns or damage to the VFS by plows or equipment for site preparation should be avoided.

Selected BMPs are described in Appendix 4.2A.

BMP Selection. The pollutant to be controlled is the major factor in BMP selection. Additional factors include the cost of the BMP and the acceptance by the farmer and the farm community. Practice selection is often times social, whereby farmers often rely on the experience of their neighbors when selecting BMPs. Practices that fit well into the operation and have high reliability and low maintenance are often preferred (see Table 4.2.7).

4.2.3 COST OF BMPs

A Division of Soil and Water Conservation (1998) cost list is provided in the references.

4.2.4 CASE STUDY

A Cost effectiveness study by Dodd and others (1995) is provided in the references.

4.2.5 REFERENCES

Athayde, D. N. P. E. Shelley, E.D. Driscoll, D. Gaboury and G. B. Boyd. 1983. "Results of the Nationwide Urban Runoff Program: Final Report," US Environmental Protection Agency. US EPA Water Planning Division, Washington, DC.

Baker, J. L. and H. P. Johnson. 1983. *Evaluating the Effectiveness of BMPs from Field Studies. In: Agricultural Management and Water Quality*. Iowa State University Press, Ames, Iowa, p. 281-305.

Baker, J. L. and J. M. Laflen. 1982. "Effects of corn residue and fertilizer management on soluble nutrient losses," *Trans. of the ASAE*, 25(2):344-348.

Baker, J.L. and J.M. Laflen. 1983. "Water quality consequences of conservation tillage," *J. Soil Water Cons*. 38(3) :186-193.

Baker, J. L., T. S. Calvin, S. J. Marley, and M. Dawelbeit. 1985. "Improved fertilizer management with a point-injector applicator," ASAE Paper No. 85-1516, ASAE, St. Joseph, MI.

Barisas, S. G., J. L. Baker, H. P. Johnson, and J. M. Laflen. 1978. "Effect of tillage systems on runoff losses of nutrients, a rainfall simulator study," *Trans. of the ASAE*, 21(5) :893-897.

Barker, J. C. and B.A . Young. 1985. Vegetative filter treatment of dairy wastewater and lot runoff in Southern Appalachia. Agricultural Waste Utilization and Management. *Proceedings of the Fifth International Symposium on Agricultural Wastes. ASAE*, St. Joseph, M.I.

Beasley, D.B. and L.F. Huggins. 1982. *ANSWERS. Users Manual*. US Environmental Protection Agency, Chicago, I.L. EPA 905/9-82-001.

Beasley, R.S. and A.B. Granillo. 1988. "Sediment and water yields from managed forests in flat coastal plain sites," *Water Res. Bull*. 24(2)361-366.

Beasley, R.S., A.B. Granillo, and V. Zillmer. 1986. "Sediment losses from forest management: Mechanical vs. chemical site preparation after clearcutting," *J. Environ. Qual*.15(4)413:416.

Beegle, D.B., P.T. Durst, and P.J. Bohn. 1989. "Farm Nutrient Management Planning Worksheet," Agronomy Series 107.Dept. of Agronomy, Pennsylvania State Univ., University Park, P.A.

Berg, R.D. 1984. "Straw residue to control furrow erosion on sloping irrigated cropland," *J. Soil and Water Cons*.58-60.

Berg, R.D. and D.L. Carter. 1980. "Furrow erosion and sediment losses on irrigated cropland," *J. Soil and Water Cons*. 35(6):267-270.

Bowles, A., G. Cashin, and J. Tippett. 1989. "An Analysis of Nutrient Loading in Lake Orange, North Carolina," Class report for FES 236, Water Quality Management. Duke University.

Brach, J. 1989. "Protecting water quality in urban areas, best management practices for Minnesota," Minnesota Pollution Control Agency. St Paul, M.N.

Brockway, C.E. 1990. University of Idaho, Boise, personal communication, August 13,1990.

Brown, G.W. 1980. *Forestry and Water Quality*. Oregon State University Bookstore, Corvallis.

Brown, M.J. 1985. "Effect of grain straw and furrow irrigation stream size on soil infiltration and infiltration," J. *Soil and Water Cons*. 40(4)389-391.

Camp, Dresser, and McKee. 1988. "Watershed Management Study: Lake Michie and Little River Watersheds," Final Report. Raleigh, NC.

Carlile, B.L. 1972. "Sediment control in Yakima Valley. Proc. of National Conference on Managing Irrigated Agriculture to Improve Water Quality," US Environmental Protection Agency and Colorado State University. p.77-82.

Carter, D.L. and R. D. Berg. 1982. “Rock Creek Intensive Monitoring Report. Attachment 3. Annual Report, Rock Creek Rural Clean Water Program,” Idaho Department of Health and Welfare, Water Quality Bureau, Boise.

Carter, D.L. and R.D. Berg. 1983. “A buried pipe system for controlling erosion and sediment loss on irrigated land,” *Soil Sci. Soc. Am. J*. 47:749-752.

Carter, D.L. and J.A. Bondurant. 1976. “Control of Sediments, Nutrients, and Adsorbed Biocides in Surface Irrigation Return Flows,” US Environmental Protection Agency EPA-600/2-76-237.

Carter, D.L., J.A. Bondurant, and C.W. Robbins. 1971. “Water soluble nitrate-nitrogen, ortho-phosphorus, and total salt balances on a large irrigation tract,” *Soil Sci. Soc. Am. Proc*. 35:331-335.

Carter, D.L., R.D. Berg, and B.J. Sanders. 1985. “The effect of furrow irrigation on crop productivity,” *Soil Sci. Soc. Am. J*. 49:207-211.

Cary, J.W. 1986. “Irrigating row crops from sod furrows to reduce erosion,” *Soil Sci. Soc. Am. J.* 50:1299-1302.

Chapra, S.C., and K.H. Reckhow. 1983. *Engineering Approaches for Lake Management, Volume II: Mechanistic Modeling*. Butterworths Pub. Boston.

Chow, V.T., D.R. Maidment, and L.R. Mays. 1988. *Applied Hydrology*. McGraw-Hill, N.Y. 572pp.

Clarke, C.D. and P.G. Waldo. 1986. “Sediment yield from small and medium watersheds,” *Proceedings of the Fourth Interagency Sedimentation Conference*. Vol 1. , Las Vegas, N.V. 3, 19-28.

Clausen, J.C. and D.W. Meals, Jr. 1989. “Water quality achievable with agricultural best management practices,” *J. Soil and Wat. Cons*. 593-596.

Clausen, J.C. and G.D. Johnson. 1990. “Lake level influences and nutrient retention in a lakeside wetland,” *J. Environ. Qual*. 19:83-88.

Coffey, S.W., W.S. Berryhill, M.D. Smolen, and D. W. Miller. 1989. “Watershed screening for nonpoint source impacts and controls,” North Carolina State University draft report to the USEPA. Washington, DC.

Coffey, S.W. and M.D. Smolen. 1990. “Effectiveness monitoring. In: Surface Water Program Guidance,” Office of Policy Planning and Evaluation. US Environmental Protection Agency, Washington, D.C.

Cook, G.D., W.T. Dickenson, and R.P. Rudra. 1985. "GAMES The Guelph Model for Evaluating the Effects of Agricultural Managements on Erosion and Sedimentation," School of Engineering, Ontario Agricultural College, University of Guelph, Ontario, Canada. Technical Report 126-71.

Davenport, T.E., Sundmaker, H and L. Austin. 1984. "Field modelling in the Highland Silver Lake Watershed," Illinois Environmental Protection Agency. IEPA/WPC/84-026.

Dean, J.D. 1983. "Potency Factors and Loading Functions for Predicting Agricultural Nonpoint Source Pollution," In: *Agricultural Management and Water Quality*. Iowa State Univ. Press, Ames, Iowa, p. 155-177.

DelVecchio, J.R. and W.G. Knisel. 1982. "Application of a Field-Scale Nonpoint Source Pollution Model," In: *Proceeding of the specialty conference on Environmentally Sound Water and Soil Management. Irrigation and Drainage Division*. ASCE, New York, N.Y.

Dennis, J., J. Noel, D. Miller, and C. Eliot. 1989. "Phosphorus control in lake watersheds: a technical guide to evaluating new development," Maine Department of Environmental Protection. Augusta, ME.

Dickey, E.C. and D.H. Vanderholm. "Vegetative filter treatment of livestock feedlot runoff," *J. Environ. Qual*. 10:27-284.

Dillaha, T.A., S.D. Lee, V.0. Shanholtz, S. Mostaghimi, and W.L. Magette. 1986. "Use of Vegetative Filter Strips to Minimize Sediment and Phosphorus Losses form Feedlots: Phase I, Experimental Plot Studies," VPI-VWRRC-Bulletin151.Blacksburg, V.A.

Dillaha, T.A., S. Mostaghimi, and C.D. Heatwole. 1988. "Tillage effects on nutrient loadings of waterways," *1988 Southern Conservation Tillage Conference Proceedings*, Special Bulletin 88-1, Tupelo, M.S. p.83-85.

Dornbush J.N., J.R. Anderson, and L.L. Harms. 1974. "Quantification of Pollutants in Agricultural Runoff," EPA-660/2-74-005, US Environmental Protection Agency, Washington, D.C.

Doyle, R.C. G.C. Stanton, and D.C. Wolf. 1977. "Effectiveness of forest and grass buffer strips in improving the water quality of manured polluted runoff," ASAE paper 77-2501.St. Joseph, MI.

Dunne, T. and L.B. Leopold. 1978. *Water in Environmental Planning*. W.H. Freeman and Co., San Francisco, C.A. 818pp.

Edwards, M.W., L.B. Owens, and, R.K. White. 1983. "Managing runoff from a small, paved feedlot," *J, Environ. Qual.*12:281-286.

Edwards, W.M., E.C. Simpson, and M.H. Frere. 1972. “Nutrient content of barnyard runoff water,” *J. Env. Qual*.1(4) :401-405.

Environmental Protection Agency, EPA. 1989. “Urban Targeting and BMP Selection. An Information and Guidance Manual for State NPS Program Staff Engineers and Managers,” Final Report. EPA Contract No. 68-C8-0034. Work Assignment No. 0-1, Amendment No. 1. Region IV, Atlanta, G.A.

Fail, Jr. J.L. B.L. Haines, and R.L. Todd. 1987. “Riparian forest communities and their role in nutrient conservation in an agricultural watershed,” *Am. J. of Altern. Ag.* 2(3):114-121.

Fitzsimmons, D.W., G.C. Lewis, D.V. Naylor, and J.R. Busch. 1972. “Nitrogen phosphorus, and other inorganic materials in waters in a gravity-irrigated area,” *Trans. ASAE.* 15:292-295.

Flanagan, D.C., G.R. Foster, W.H. Neibling, and J.P. Burt. “Simplified equations for filters strip design,” *Trans. ASAE*. 32(6) :2001-2007.

Fowler, W.B., T.A. Anderson, and J.D Helvey. 1988. “Changes in water quality and climate after forest harvest in central Washington state,” Research Paper PNW-RP-388, USDA Forest Service, Pacific Northwest Research Station, 12 pp.

Gakstatter, J.H., M.O. Allum, S.E. Dominquez, and M.R. Crouse. 1978. “A survey of phosphorus and nitrogen levels in treated municipal wastewaters,” *J. Wat. Poll. Cont. Fed*. 50(4):718-722.

Gale, J. A., Line, D. E., Osmond, D. L., Coffey, S. W., Spooner, J., Arnold, J. A., Hoban, T. J., and Wimberley, R. C. 1993. “Evaluation of the Experimental Rural Clean Water Program,” North Carolina Cooperative Extension Service, North Carolina State University, Raleigh, NC, under a grant from the Nonpoint Source Control Branch, US Environmental Protection Agency, Washington, DC.

Gilbertson, C.B., F.A. Norstadt, A.C. Mathers, R.F. Holt, A.P. Barnett, T.M. McCalla, C.A. Onstadt, and R.A. Young. 1979. “Animal Waste Utilization on Cropland and Pasture,” EPA-600/2-79-059, USEPA, Washington D.C.

Glasser, S.P. 1989. “Summary of water quality effects from forest practices in the South,” USDA Forest Service Southern Region. Available from SCS, South National Technical Center, Fort Worth, T.X.

Haith, DA. and R.A. Loehr (eds). 1979. “Effectiveness of Soil and Water Conservation Practices for Pollution Control,” EPA-600/3-79-106. US Environmental Protection Agency, Athens, G.A.

Haith, D.A. and Shoemaker, L.L. 1987. “Generalized Watershed Loading Functions for Streamflow Nutrients,” *Water. Res. Bull*. 12(2) :471-478.

Haith, D.A. and Tubbs, L.J. 1981. “Watershed loading functions for nonpoint sources,” *J. Env. Eng. Div. ASCE* 107 (EEl) :121-137.

Hamon, W.R. 1961. “Estimating potential evapotranspiration,” *Proceedings of the American Society of Civil Engineers, J. Hydraul. Div*. 87(HY3):107-120.

Hamlett, J.M., J.L. Baker, and H.P. Johnson.1987. “Size distribution of sediment transported within and from small agricultural watersheds,” *Trans. ASAE* 30(4):998-1004.

Harmon, J.D., E.R. Collins, Jr., and F.B. Givens. 1986. “Animal Waste Utilization Software User's Guide,” Virginia Polytechnic and State Univ., Blacksburg, V.A.

Hewlett, J.D. 1982. *Principles of Forest Hydrology*. University of Georgia Press, Athens.

Hope Henning, M. and K.H. Reckhow. 1990. “Guidance manual for EUTROMOD spreadsheet software,” Draft report. Duke University .

Israelson, O.W., G.D. Clyde, and C.W. Lauritzen. 1946. “Soil Erosion in Small Irrigation Furrows,” Utah Agricultural Experiment Station Bulletin 320.

Johnson, A.H. 1975. “Phosphorus Export from the Fall Creek Watershed,” Ph.D. Thesis, Cornell Univ., University Microfilms, Ann Arbor Michigan.

Keeney, R.L. 1977. “A utility function for examining policy affecting salmon in the Skeena River,” *J. Fish. Res. Board Canada*. 34:49-63.

Keeney, R.L., and H. Raiffa. 1976. *Decisions with Multiple Objectives*. Wiley. New York.

Kemper, W.D., T.J. Trout, M.J. Brown, and R.C. Rosenau. 1985. “Furrow Erosion and Water and Soil Management,” *Trans. of the ASAE* 28(5):1564-1572.

King, L.G., B.L. McNeal, F.A. Ziari, and S.C. Matulich.1984. “On-Farm Improvements to Reduce Sediment and Nutrients in Irrigation Return Flow,” US Environmental Protection Agency, Ada, O.K. EPA-600/2-84-044.

Knisel, W.G. 1980. “CREAMS: A Field-Scale Model for Chemicals, Runoff, and Erosion from Agricultural Management Systems,” USDA Sci. and Educ. Adm. Conser. Res. Rep. No. 26.

Lawrence, I. 1986. “Source and Fate of Urban Runoff Constituents and Their Management,” *Proc. of 12thSymposium on Stormwater Quality in Urban Areas*, Water Research Foundation of Australia, Univ. of Woollongong.

Likens, G.E., F.H. Bormann, R.S. Pierce, N.M. Johnson, D.W. Fisher, and R.S. Pierce. 1970. "Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem," *Ecol. Monogr*. 40:23-47.

Lynch, J.A., E.S. Corbett and K. Mussallem. 1985. "Best management practices for controlling nonpoint-source pollution on forested watersheds," *J. Soil Water Cons*.40(1) :164-167.

Lynch, J.A. and E.S. Corbett. 1990. "Evaluation of best management practices for controlling nonpoint pollution from silvicultural operations," *Water. Res. Bull*.26(1) :41-52.

Maas, R.P., M.D. Smolen, C.A. Jamieson, and A.C. Weinburg. 1987. *Setting Priorities: The Key to Nonpoint Source Control*. EPA Office of Water Regulations and Standards, Washington, DC 20460.

Maner, S.B. 1958. "Factors affecting sediment delivery rates in the Red Hills physiographic area," *Trans. Am. Geophys. Union*. 39(4):669-675.

Martin, C.W. and R.S. Pierce. 1980. "Clearcutting patterns affect nitrate and calcium in streams in New Hampshire," *J. Forestry* 78(5):268-272.

Martin, C.W., D.S. Noel, and C.A. Federer. 1984. "Effects of forest clearcutting in New England on stream chemistry," *J. Environ. Qual*. 13(2):204-210.

McClurkin, D., P. Duffy S. Ursic and N. Nelson. 1985. "Water quality effects of clearcutting Upper Coastal Plain loblolly pine plantations," *J. Environ. Qual*. 14(3).

McElroy, A.D. S.Y. Chin, J.W. Nebgen, A. Aleti, F.W. Bennett. 1976. "Loading Functions for Assessment of Water Pollution from Nonpoint Sources," EPA-600/2-76-151, US EPA, Washington, D.C.

McNeal, B.L., N.K. Whittlesey, and V.F. Obersinner. 1982. "Appendix B: Sediment and Nutrient Loss Estimates From Irrigated Agriculture," In: *Robillard, P.D., M.F. Walter, and L.M. Bruckner. Planning Guide for Evaluating Agricultural Nonpoint Source Controls*. USEPA, Athens, G.A. EPA 600/3-82-021.

Meals, D.W. 1987. "LaPlatte River Watershed Water Quality Monitoring and Analysis Program. Report No. 9," Vermont Water Resources Research Center. Univ. Of Vermont, Burlington.

Megahan, W.F. and P.N. King. 1985. "Identification of critical areas in forest lands for controlling nonpoint sources of pollution on forested watersheds," *Environ. Man*. 9(1):7-18.

Meta Systems, Inc. 1973. "Effluent Charges," Prepared for the USEPA, Contract No. 68-01-0566.

Metropolitan Washington Council of Governments, (MWCG) .1978. "Land Use/Runoff Quality Relationships in the Washington Metropolitan Area, Occoquan/Four Mile Run Non-Point Source Correlation Study," Prepared by the Northern Virginia Planning District Commission and the Dept. of Civil Engineering, Virginia Polytechnic Institute and State University.

Miller, D.E., J.S. Asrstad, and R.G. Evans. 1987. "Control of furrow erosion with ceop residues and surge irrigation," *Soil Sci. Soc. Am. J.* 51:421-425.

Miller, E. L. 1984. "Sediment yield and storm flow response to clear-cut harvest and site preparation in the Ouachita Mountains," *Wat. Res. Res*. 20(4):471-475.

Miller, E.L. R.S. Beasley, and E.R. Lawson. 1988. "Forest harvest and site preparation effects on erosion and sedimentation in the Ouachita Mountains," *J. Env ir. Qual*. 17(2):219-225.

Mills, W.B., D.B. Porcella, M.J. Ungs, S.A. Gherini, K.V. Summers, L. Mok, G.L. Rupp, G.L. Bowie, and D.A. Haith.1985. "Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water-Part I (Revised 1985)," EPA-600/6-85-002a US EPA, Washington, D.C.

Miner, J.R., J.K. Koelliker, and M.J. English. 1981. "Predicting cattle feedlot runoff and retention basin quality," USEPA Project Summary, Ada, O.K. EPA-600/52-80-192.

Montgomery, R.H., J.T. Acre, P. De Curnou, E.D. Driscoll, and E.W. Strecker. 1989. "A Probabilistic methodology for assessing impacts of urban runoff on lake water quality," Draft report prepared by Woodward-Clyde Consultants for the USEPA. Washington, DC.

Moore, D.G. 1975. "Impact of forest fertilization on water quality in the douglas-fir region: A summary of monitoring studies," *Proceedings, 1974 National Convention Society of American Foresters*, New York City, September 22-26. p. 209-219.

Morton, T.G., A.J. Gold, and W.M. Sullivan. 1988. *J. Environ. Qual*. 17(1):124-130.

NALMS. 1988. *Lake and reservoir restoration guidance manual*. North American Lake Management Society report prepared for the USEPA. Washington, DC.

National Water Quality Evaluation Project (NWQEP) . 1982a. "Best Management Practices for Agricultural Nonpoint Source Control. I. Animal Waste. Rural Nonpoint Source Control Water Quality Evaluation and Technical Assistance," North Carolina Agricultural Extension Service, Biological and Agricultural Engineering Department, North Carolina State Univ., Raleigh, N.C.

National Water Quality Evaluation Project (NWQEP). 1982b. "Best Management Practices for Agricultural Nonpoint Source Control. II. Commercial Fertilizer. Rural Nonpoint Source Control Water Quality Evaluation and Technical Assistance," North Carolina Agricultural Extension Service, Biological and Agricultural Engineering Department, North Carolina State Univ.,Raleigh, N.C.

National Water Quality Evaluation Project (NWQEP). 1988. "NWQEP 1987 Annual Report: Status of Agricultural Nonpoint Source Projects," National Water Quality Evaluation Project, North Carolina Agricultural Extension Service, North Carolina State Univ., Raleigh, N.C.

National Water Quality Evaluation Project (NWQEP). 1989. "NWQEP 1988 Annual Report: Status of Agricultural Nonpoint Source Projects," National Water Quality Evaluation Project, North Carolina Agricultural Extension Service, North Carolina State Univ., Raleigh.

Naylor D.V. and J.R. Rauch. 1973. "Effects of irrigation, fertilization, and other cultural practices on water quality," *Res. Tech. Rep*., Idaho Water Resources. Inst.

NCDWQ. 1996. "Neuse River Nutrient Sensitive Waters Management Strategy Rules," North Carolina Administrative Code 15A, NCAC 2B.0202

Newell, A.D., L.C. Stanley, M.D. Smolen, and R.P. Maas. 1986. "Overview and Evaluation 108-A Great Lakes Demonstration Projects," North Carolina Agricultural Extension Service, Raleigh, NC. EPA Great Lakes Program Office, EPA-905/9-86-006, 66pp.

New Jersey State Soil Conservation Committee. 1982. Standards for Soil Erosion and Sediment Control in New Jersey. N.J. Department of Agriculture, Trenton, N.J.

Nutter, W.L. and J.W. Gaskin. 1989. "Role of streamside management zones in controlling discharges to wetlands," In: *Proceedings of the Symposium: The forested Wetlands of the Southern United States*, July 12-14, 1988, Orlando, F.L. General Technical Report SE-SO. Asheville, N.C., U.S. Department of Agriculture, S.E. Forest Experiment Station.

Occoquan Watershed Monitoring Laboratory (OWML). 1983. "Final Contract Report: Washington Area NURP Project," prepared for Metropolitan Washington Council of Governments. Manassas, Virginia. 45Q pp.

Omernik, J. M. 1977. "Nonpoint Source-Stream Nutrient Level Relationships: Ad Nationwide Stud," EPA 600/3-77-105. USEPA, Corvallis, O.R.

Overcash, M.R., F.J. Humenik, P.W. Westerman, D.M. Covil, and J.W. Gilliam. 1976. "Overland flow pretreatment of poultry manure," ASAE paper 76-4517 St. Joseph, M.I.

Phillips, B.C. and A.G. Goyen. 1987. "Guidelines for Improving Urban Stormwater Quality Using Water Quality Control Structures," *Proc. Seminar on Urban Runoff Water Quality*, I.E. Australia/A.W.W.A., Sydney.

Phillips, R.L. 1981. "Maps of runoff volumes from feedlots in the United States. Livestock Waste: A renewable resource," *Proc. 4th International Symposium on Livestock Wastes-1980.* April 15-17 1980, Amarillo, T.X. Published by the American Society of Agricultural Engineering, St, Joseph, M.I.

Pierce, R.S., J.W. Hombeck, G.E. Likens, and F.H. Bormann. 1970. "Effect of elimination of vegetation on stream water quantity and quality," IAHS-AISH Publ. 96:311-328.

Reckhow, K.H. 1990. *EUTROMOD Spreadsheet Program, A regional modeling scheme for nutrient runoff and lake trophic state modeling*. Duke University.

Reckhow, K.H., M. N. Beaulac, and J.T. Simpson. 1980. "Modeling phosphorus loading and lake response under uncertainty: a manual and compilation of export coefficients," USEPA. 440/5-80-011. Washington, DC.

Reckhow, K.H. and S.C. Chapra. 1983. *Engineering Approaches for Lake Management, Volume I: Data Analysis and Empirical Modeling*. Butterworth Publishers. Boston.

Reckhow, K.H., J.P. Hartigan, and S.W. Coffey. 1989. "Lake nutrient budget development for state level applications," In: *Proceedings of a National Conference on Enhancing States' Lake Management Programs.* North American Lake Management Society. Washington, DC.

Reckhow, K.H., K. Kepford, and W. Warren-Hicks. 1990. "Methods for the detection of water quality trends in lakes," Draft Report to the USEPA, Washington, D.C.

Reckhow, K.H., and T.B. Stockton, Jr. 1990. "Regional Lake Eutrophication Models. Draft Report to the USEPA," Washington, D.C.

Reckhow, K.H., and C. Stow. 1990. *Monitoring design and data analysis for trend detection. Lake and Reservoir Management (in press)* .

Riekerk, H. 1983. "Impacts of silviculture on flatwoods runoff, water quality and nutrient budgets," *Water Resources Bull*. 19(1):73-79.

Riekerk, H., D.G. Neary, and W.T. Swank. 1989. "The magnitude of upland silvicultural nonpoint source pollution in the south," USDA Forest Service, Southeast Forest Experiment Station, Asheville N.C., Gen. Tech. Rep. 50, pp 8-18.

Ritter, W.F., A.E.M. Chirnside, and R.W. Lake. 1989. "Influence of Best Management Practices on Water Quality in the Appoquinimink Watershed," J. *Environ. Sci. Health*, A24(8) :897-924.

Robillard, P.D., M.F. Walter, and M. Hiraizumi. 1983. "Phosphorus control alternatives for dairy barnyards," ASAE Technical paper 83-2546.

Roehl, J.W. 1962. "Sediment source areas, delivery ratios, and influencing morphological factors," *International Association of Scientific Hydrology*. 59:202-213. Symposium of Bari, October 1-8, 1962.

Soil Conservation Service, SCS. 1983. "National Engineering Handbook, Section 3 Sedimentation," Second edition.

Soil Conservation Service, SCS. 1987. "Practice Standards for Filter Strips. National Handbook of Conservation Practices," US Department of Agriculture, Washington, D.C.

Soil Conservation Service (SCS) 1986. "Urban Hydrology for Small Watersheds Manual," Washington, D.C., Technical Release 55. Second Edition. P0 87-101580/AS A08.

SCS 1988. "Water Quality Technical Workshop: Integrating Water Quality and Quantity into Conservation Planning," October 24-27, 1988, Raleigh, NC.

Schueler, T.R. 1987. "Controlling Urban Runoff: A Practical Manual for Planning and Designing Urban BMPs," Metropolitan Washington Council of Governments.

Schulte, E.E., K.A. Kelling, and S.M. Combs. U"ser Guide to Nutrient Adjustment Worksheet," 15 June 1989, Soil Science Department, University of Wisconsin, Madison, WI 53706.

Shapiro, J. and H. Pfannkuck. 1973. "Interim Report No. 9," Limnological Research Center University of Minnesota. St. Paul, Minnesota.

Smith, G.E., F.D. Witaker, and H.G. Heinemann. 1974. "Losses of fertilizers and pesticides from claypan soils," US Environmental Protection Agency, EPA 660/2-74-068. Washington, D.C.

Smolen, M.D. 1988. "Targeting critical water quality areas," In: *Rural Clean Water Program 1988 Workshop Proceedings*. September 12-15, St. Paul M.N., NWQEP, Agricultural Extension Service, North Carolina State University, Raleigh, N.C.

Smolen, M.D., D.W. Miller, L.C. Wyatt, J, Lichthardt, A.L. Lanier, and R.G. Jessup. 1988. "Erosion and Sediment Control Planning and Design Manual," State of North Carolina, North Carolina Sediment

Control Commission. North Carolina Department of Health, Environment and Natural Resources, Raleigh.

Stewart, B.A., D.A. Woolhiser, W.H. Wischmeier, J.H. Caro, and M.H. Frere. 1975. "Control of Water Pollution from Croplands, Volume I: A manual for guideline development," EPA-600/2-75-026, US EPA, Washington, D.C.

Stewart, B.A., D.A. Woolhiser, W.H. Wischmeier, J.H. Caro, and M.H. Frere.1976. "Control of Water Pollution from Croplands, Volume II: An overview," EPA-600/2-75-026b, US EPA, Washington, D.C.

Stahre, P. and B. Urbonas. 1990. *Stormwater Detention: For Drainage, Water Quality, and CSO Management*. Prentice Hall, Englewood Cliffs, NJ, 338 pp.

Schwer, C.B. and J.C. Clausen. 1989. "Vegetative Filter Treatment of Dairy Milkhouse Wastewater," *J. Environ. Qual*. 18:446-451.

Thompson, D.B., L.T. Loudon, and J.B. Gerrish. 1979. "Animal manure movement in winter runoff for different surface conditions," In: *Best Management Practices for Agriculture and Silviculture*. Ann Arbor Science, Ann Arbor, M.I.

Timmons, D.R., R.E. Burwell, and R.F. Holt. 1973. "Nitrogen and phosphorus losses in surface runoff from agricultural land as influenced by placement of broadcast fertilizer," *Water Resources Res*. 9:658-67.

Tubbs L.J. and D.A. Haith. 1977. "Simulation of nutrient losses from cropland," ASAE Paper. 77-2502. American Society of Agricultural Engineers. St. Joseph, M.I.

University of Wisconsin-Extension (UWEX). 1989. "Nutrient and Pesticide Best Management Practices for Wisconsin Farms," Agricultural Bulletin Building, 30 North Murray Street, Madison, WI 53706.

US Environmental Protection Agency (EPA) 1974. "Nitrogen and phosphorus in wastewater effluents," Working Paper. No 22 National Eutrophication Survey. Pacific Northwest Environmental Research Laboratory, Corvallis Oregon, GPO 697-032.

US Geological Survey (USGS). 1985. "National Water Summary 1985-Hydrologic events and surface-water resources," US Geological Survey. Water-Supply Paper 2300.

van der Valk, A.G., C.B. Davis, J.L. Baker, and C.E. Beer. 1979. “Natural fresh water wetlands as nitrogen and phosphorus traps for land runoff,” p. 57-467 In: *P.E. Greeson et al. (ed.) Wetland functions and values: The state of our understanding. Am. Wat. Res. Assoc*. Minneapolis, M.N.

Vitosh, M.L. and L.K. Carroll. 1987. “Fertilizer Recommendations. An MSU Fertilizer Recommendation Software Program,” Cooperative Extension Service, Michigan State University, E. Lansing, M.I.

WDATCP, Wisconsin Department of Agriculture, Trade, and Consumer Protection. 1989. “Nutrient and Pesticide Best Management Practices for Wisconsin Farms,” Univ. Of Wis. Extension, Madison.

Waldo, P.G. Sediment yields from large watersheds. “River Sedimentation,” *Volume III. Proceedings of the Third International Symposium on River Sedimentation*, School of Engineering. Univ. of Mississippi, University, M.S.

Walker, W.W., Jr. 1986. “Empirical Methods for Predicting Eutrophication in Impoundments; Report 4, Phase III: Applications Manual,” Technical Report E-81-9. US Army Engineer Waterways Experiment Station. Vicksburg, MS.

Walker, W.W., Jr. 1988. “P8 Urban Catchment Model. Program Documentation. Version 1.1,” prepared for EIP, Inc., 6 Maple Street, Northborough, M.A. and Narrangansett Bay Project, Providence, R.I.

Wall, D.B. S.A. McGuire, and J.A. Magner. 1989. “Water Quality Monitoring and Assessment in the Garvin Brook Rural Clean Water Project Area,” Minnesota Pollution Control Agency, Division of Water Quality, St. Paul.

Walter, M.F., R.C. Loehr, R.J. Linkenheil, D.F Sherman, B.K. Richards, and P.D. Robillard. 1983. “Evaluation of Milhouse Wastewater Treatment. Agricultural Engineering,” Cornell Univ, Ithaca, N.Y.

Walter, M.F., T.L. Richard, P.D. Robillard, and R. Muck.1987. “Manure Management with Conservation Tillage,” In: *Logan, T.J., J.M. Davidson, J.L. Baker and M.R. Overcash. eds. Effects of Conservation Tillage on Groundwater*. Lewis Publishers, Chelsea, M.I.

Water Resources Agency for New Castle County (WRANCC) . “Appoquinimink River Basin Project Rural Clean Water Program,” Final Report. 1986. Delaware.

Woodward-Clyde Consultants. 1989. “Synoptic analysis of selected rainfall gages throughout the United States. Draft report for the USEPA,” EPA WA-10, 8910148B.1100. Oakland, CA.

Willrich, T.L and J.O. Boda. 1976. “Overland flow treatment fo swinelagoon effluent,” ASAE paper 76-4515, St. Joseph, M.I.

Williams, J.R., A.D. Nicks, and J.G. Arnold. 1985. “Simulator for water resources in rural basins,” *J. Hydraulic. Eng*.111(6) :970-986.

Winter, T.C. 1981. “Uncertainties in estimating the water balance of lakes,” *Water Resourc. Bulletin*. 17(1):82-115.

Wischmeier, W.H. and Smith, D.D. 1978. “Predicting Rainfall Erosion Losses--A Guide to Conservation Planning,” Agricultural Handbook 537, US Dept of Agriculture, Washington, D.C.

Yang, S.Y., J.H. Jones, F.J. Olsen, and J.J. Peterson. 1980. “Soil as a medium for dairy liquid waste disposal,” *J. Environ. Qual*. 9(3):370-372.

Young, R.A., M.A. Otterby, and A. Roos. 1982. “An Evaluation System to Rate Feedlot Pollution Potential. Agricultural Research Service,” ARM-NC-17. US Department of Agriculture, Peoria, I.L.

Young, R.A., C.A. Onstad, D.D. Bosch and W.P. Anderson.1987. “AGNPS: Agricultural Non-Point-Source Pollution Model, A watershed analysis tool,” US Department of Agriculture. Conservation Research Report, 35. 80 pp.

Young, R.A., C.A. Onstad, D.D. Bosch, and W.P. Anderson. 1994. *Agricultural Nonpoint Source Pollution Model, Version 4.03 AGNPS User’s Guide*. USDA Agricultural Research Service, Morris, MN.

Table 4.2.1. Total Nitrogen and Total Phosphorus Concentration in Wastewater Treatment Plant Effluent		
Treatment Type	Mean Nitrogen (mg/l)	Mean Phosphorus (mg/l)
Activated Sludge	15.8	5.91
Trickling Filter	17.9	7.25
Phosphorus Removal	4	2.85
Primary Settling and Digestion	23.8	8.71
Oxidation Pond	17.1	6.42
Sand Filter	--	5.02
(after Gakstatter et al. 1978 and Reckhow and Chapra 1983)		

Table 4.2.2. Lot Sizes, Percent Impervious Area and Nutrient Export for Various Land Uses		
Land Use	Lot Size (ha)	Percent Impervious
Rural Residential	2.0	4
Large Residential	0.8	6
Low Density Residential	0.4	12
Low-medium Density Residential	0.2	20
Medium Density Residential	0.1	25
(Adapted from Camp, Dresser and McKee 1988)		

Table 4.2.3. Summary of Best Management Practices				
Practice	Sediment	Dissolved Nutrients	Attached Nutrients	Groundwater
Development and Urban				
Construction	3	1	3	1
Source Controls	2	2	2	1
Vegetative Controls	2	2	2	1
Infiltration Devices	3	2-3	3	--
Dry Detention Devices	3	1	1-2	1
Wet Detention Devices	3	2	3	1
Agriculture				
Conservation Tillage	3	1	3	--
Terraces	3	2	3	--
Irrigation Furrow	3	2	3	1
Animal Waste Systems	1	3	3	3
Nutrient Management Systems	1	3	3	3
Grassed Waterways	1-2	1	1-2	1
Filter Strip	1-2	1	1-2	1
Key to Ratings: 1 - no control to low effectiveness; 2 - low to medium effectiveness; 3 - medium to high effectiveness; – may increase loading in some cases. NOTE: Performance of BMPs may be lower than indicated for some sites.				
(Adapted from Schueler 1987; Brach 1989)				

Table 4.2.4. Computed Fractions of Clay, Silt and Aggregates
Entering a Grass Filter Strip from a Silt Loam Soil

Computed Fraction Entering Strip				
		Primary Clay	Primary Silt	Small Aggregate
Slope	Roughness			
Low (0.5%)	Low	0.12	0.28	0.46
	Mod	0.29	0.43	0.27
	High	0.32	0.47	0.21
Mod. (2.5%)	Low	0.05	0.24	0.36
	Mod	0.13	0.28	0.43
	High	0.28	0.42	0.28
High (5%)	Low	0.05	0.24	0.36
	Mod	0.08	0.24	0.44
	High	0.24	0.39	0.31
Steep (10%)	Low	0.05	0.24	0.36
	Mod	0.06	0.24	0.39
	High	0.21	0.35	0.34
(after Flanagan et al. 1989)				

Table 4.2.5. Estimated o Values for Five Particle Classes
of Four Soil Types

Particle Class	Soil Type			
	Sand	Silt	Silt Loam	Clay
Primary Clay	0.110	0.110	0.110	0.110
Primary Silt	2.84	2.84	2.84	2.84
Small Aggregate	13.5	13.5	13.5	95.0
Large Aggregate	643	643	954	2670
Primary Sand	820	820	820	820
(after Flanagan et al. 1989)				

Table 4.2.6. Sample Calculations for the Sediment Delivery Ratio

Particle Class	$x_u*\phi$		f_i		SD
primary clay	0.998	x	0.05	=	0.05
primary silt	0.994	x	0.24	=	0.23
primary aggregate	0.761	x	0.36	=	0.27
large aggregate	0.000	x	--	=	0.0
primary sand	0.000	x	--	=	0.0

Table 4.2.7 NRCS Conservation Practices, Pollutants Potentially Controlled, and Sources of Pollutants (Cite)													
		POLLUTANT TO BE CONTROLLED											
NRCS CODE	PRACTICE	TP	PP	OP	TN	TKN	ON	NI	AM	SE	BOD	FC	SOURCE
560	Access Road									**			CD, CI, PA, RG, FO
575	Animal Trails and Walkways		**						**	**		**	PA, RG
310	Bedding	**			**					**			CD, CI
314	Brush Management									**			RG
317	Compost facility	**			**		**		**			**	CA
327	Conservation Cover	**	**	**	**					**			CD, CI
322	Channel Vegetation	**	**	**		**		**	**	**			CD, CI, PA, RG, GU
324	Chisel Tillage	**	**	**	**	**	**	**	**	**			CD, CI
328	Conservation Crop Rotation	**	**	**	**	**	**	**	**	**			CD, CI
330	Contour Farm	**	**	**	**	**	**	**	**	**		**	CD
335	Controlled Drainage	**	**	**	**	**	**	**		**			CD, CI
340	Cover and Green Manure Crop	**	**		**					**			CD, CI
342	Critical Area Planting	**	**		**	**	**		**	**		**	GU, CD, PA, RG
352	Deferred Grazing		**	**	**	**		**	**	**		**	PA, RG
362	Diversion	**	**			**			**	**			CD, CI, CA
382	Fencing	**	**	**	**	**	**		**	**			CA, PA, RG
386	Field Border	**	**				**			**		**	CD, CI
393A	Filter Strips	**	**		**	**	**	**	**	**		**	CA, CD, CI, FO, PA, RG, CONSTR
400	Floodway Diversion												
410	Grade Stabilization Structure		**							**			GU, ST
412	Grass Waterway	**	**		**	**			**	**			CD, CI
548	Grazing Land Mechanical Treatment	**			**	**		**		**			PA, RG
561	Heavy Use Area Protection	**			**					**		**	CA, PA, RG
422	Hedgerow Planting	**			**					**			CA, PA

Table 4.2.7 (Continued)													
		POLLUTANT TO BE CONTROLLED											
NRCS CODE	PRACTICE	TP	PP	OP	TN	TKN	ON	NI	AM	SE	BOD	FC	SOURCE
423	Hillside Ditch	**			**					**			GU
320	Irrigation Canal or Lateral	**		**	**					**			CI
388	Irrigation Field Ditch	**		**	**					**			CI
464	Irrigation Land Leveling	**		**	**					**			CI
552A	Irrigation Pit or Regulating Reservoir, Irrigation Pit	**		**	**					**			CI
552B	Irrigation Pit or Regulating Reservoir, Regulating Reservoir	**		**	**					**			CI
436	Irrigation Storage Reservoir	**		**	**					**			CI
443	Irrigation System, Surface and Subsurface	**		**	**					**			CI, PA
441	Irrigation System, Trickle	**		**	**					**			CI
428B	Irrigation Water Conveyance, Ditch and Canal Lining, Flexible Membrane	**		**	**					**			CI
428C	Irrigation Water Conveyance, Ditch and Canal Lining, Galvanized Steel	**		**	**					**			CI
428A	Irrigation Water Conveyance, Ditch and Canal Lining, Nonreinforced Concrete	**		**	**					**			CI
430AA	Irrigation Water Conveyance, Pipeline, Aluminum Tubing	**		**	**					**			CI
430DD	Irrigation Water Conveyance, Pipeline, High-Pressure, Underground, Plastic	**		**	**					**			CI
430EE	Irrigation Water Conveyance, Pipeline, Low-Pressure, Underground, Plastic	**		**	**					**			CI
430CC	Irrigation Water Conveyance, Pipeline, Nonreinforced Concrete	**		**	**					**			CI

Table 4.2.7 (Continued)													
		POLLUTANT TO BE CONTROLLED											
NRCS CODE	PRACTICE	TP	PP	OP	TN	TKN	ON	NI	AM	SE	BOD	FC	SOURCE
430HH	Irrigation Water Conveyance, Pipeline, Rigid Gated Pipeline	**		**	**					**			CI
430FF	Irrigation Water Con-veyance, Pipeline, Steel	**		**	**					**			CI
447	Irrigation System, Tailwater Recovery	**		**	**					**			CI
449	Irrigation Water Management									**			CI, PA
472	Livestock Exclusion	**	**	**	**	**	**		**	**		**	PA, RG
482	Mole Drain	**		**	**			**		**			CA
484	Mulching	**	**	**	**	**			**	**			CD, CI
329	No Till	**	**	**	**	**	**	**	**	**			CD, CI
590	Nutrient Management	**	**	**	**	**	**	**	**				CA, CI, CD
512	Pasture and Hay Planting	**	**		**				**	**		**	CD, PA
378	Pond	**	**		**	**	**	**		**			CD, CI, RG, CONSTR, PA
462	Precision Land Forming									**			CD, CI
528A	Prescribed Grazing		**							**			RG, PA
532	Pumped Well Drain				**								CA, CI
550	Range Planting	**								**			RG
554	Regulating Water in Drainage Systems	**	**	**	**					**			CI, CA
344	Residue Management, Seasonal	**	**	**		**	**	**	**	**			CD, CI
391A	Riparian Forest Buffer	**	**		**	**	**	**	**	**		**	CA, CD
555	Rock Barrier	**								**			CD
558	Roof Runoff Management	**			**					**		**	CA
570	Runoff Management System	**	**	**	**	**		**	**	**			CONST

Table 4.2.7 (Concluded)													
		POLLUTANT TO BE CONTROLLED											
NRCS CODE	PRACTICE	TP	PP	OP	TN	TKN	ON	NI	AM	SE	BOD	FC	SOURCE
350	Sediment Basin	**			**	**			**	**			CD, CI, CA, CONSTR
574	Spring Development												NA
442	Irrigation System, Sprinkler												CI
580	Streambank & Shoreline Protection	**	**			**			**	**			STREAM-BANKS
585	Stripcropping, Contour	**	**	**	**	**		**	**	**		**	CD, CI
587	Structure for Water Control	**	**		**	**		**		**			CI, GU
586	Stripcropping, Field	**	**	**	**	**			**	**		**	CD
606	Subsurface Drain	**	**	**	**			**	**	**		**	CD, CI
607	Surface Drainage, Field Ditch	**	**		**	**		**	**	**		**	CD, CI
608	Surface Drainage, Main or Lateral	**			**			**				**	CD, CI
600	Terrace	**	**		**	**	**		**	**			CD, CI
612	Tree Shrub Establishment	**	**		**	**		**	**	**			CD, CI, FO
614	Trough or Tank									**			
312	Waste Management System	**	**	**	**	**	**	**	**	**		**	CA
313	Waste Storage Facility	**	**	**	**	**	**	**	**	**		**	CA
359	Waste Treatment Lagoon	**	**	**	**	**	**	**	**			**	CA, PA
633	Waste Utilization	**	**	**	**	**	**	**	**				CA, CD, CI, PA
638	Water & Sediment Control Basin	**	**	**	**	**	**	**		**			CD, CA, PA, RG, GU, IR
640	Waterspreading	**			**					**			CA, CI, PA, RG

POLLUTANTS: TP = Total Phosphorus; PP = Particulate Phosphorus; OP = Orthophosphate; TN = Total Nitrogen; TKN = Kjeldahl Nitrogen; ON = Organic Nitrogen; NI = Nitrate; AM = Ammonia; SE = Sediment; BOD = Biological Oxygen Demand; FC = Fecal Californ

POLLUTANT SOURCES: CA = Confined Animals; CD = Cropland (Dryland); CI = Cropland (Irrigated); Const. = Construction; FO = Forest; GU = Gullies; PA = Pasture; RG = Rangeland; NA = Not Applicable

4.3 IN-RESERVOIR TECHNIQUES: SELECTIVE WITHDRAWAL

4.3.1 MULTILEVEL SELECTIVE WITHDRAWAL

4.3.1.1 Problem Addressed

The operation of a multilevel intake structure requires the consideration of numerous project conditions and constraints, the most important of which is thermal stratification. As stratification develops, the limits of withdrawal for a given port are reduced because of the density differences imparted by the thermal stratification. Thus, flow through a port at a given elevation may not result in a release temperature similar to that observed at the center-line elevation of the port or that desired to meet a downstream objective.

Selective withdrawal, which is defined as the capability to describe the vertical distribution of withdrawal from a density-stratified reservoir and then use that capability to selectively release the quality of water that is desired, can be used to determine the appropriate or best available operation of a release structure. It can also be used in the design of multilevel intakes as well as in the modification of existing projects to achieve a given release water quality criteria.

4.3.1.2 Theory

This technique relies on manipulation of the density-impacted withdrawal pattern to control the release quality from a structure. The withdrawal pattern that is set up as the result of release of water through a port during stratified conditions is defined as the withdrawal zone (Figure 4.3.1). Water within this zone will ultimately be released through the port, even though the rate of release from individual strata of the pool varies. By identifying the portion of each layer released under a given operating condition, the release quality can be predicted.

4.3.1.3 Selective Withdrawal Methodology

Much research has been performed in the area of water withdrawal through a point sink (such as a single port) or through a linear sink (such as a line of penstock openings for hydropower). The effort has focused on the withdrawal profile that is created and on the way in which density stratification and physical boundaries affect the withdrawal limits (beyond which, water is not drawn into the port). When water is withdrawn through a single port, water from well above and below the horizontal center line of that port is withdrawn.

The development of these upper and lower withdrawal limits is dependent on the density gradient present in the water column, the flow through that port, and local geometric effects. Beyond the upper and lower limits of withdrawal, insufficient energy is available in the flow to entrain the water from these levels into the main flow (Wilhelms 1986).

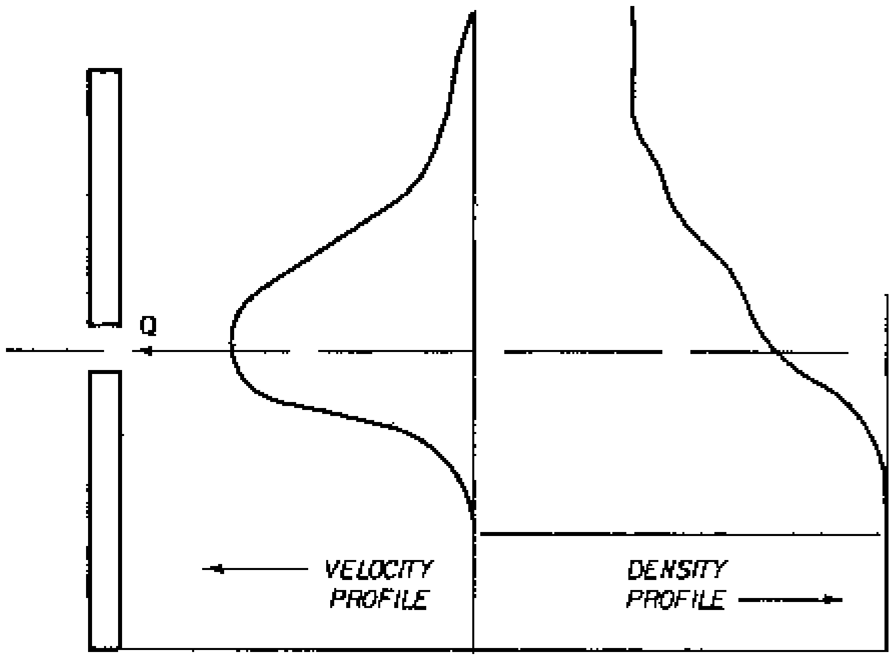


Figure 4.3.1 Schematic of withdrawal zone

The design methodology for a selective withdrawal system, given in the following paragraphs, is based on physical and numerical model investigations.

Since the density gradient in the water column has a direct influence on the potential energy of the water at various levels, it strongly affects the development of the withdrawal limits. For example, if the stratification is weak or nonexistent, the withdrawal limits may extend from the bottom to the water surface. If the water column is strongly stratified, this will cause the withdrawal limits to shrink so that the port is withdrawing from a smaller cross-sectional area. Therefore, the density stratification (usually temperature) for design conditions must first be determined.

If the selective withdrawal investigation involves an existing reservoir, existing profiles may be used. If the selective withdrawal investigation is for a reservoir not yet constructed, a one-dimensional numerical model can be used to predict typical temperature profiles. Additional information relative to the selective withdrawal investigation, such as local topography and operating criteria (hydraulic and water quality), should also be compiled.

The development of the withdrawal zone often is modified by a physical boundary. The port may be too close to the top or bottom of the reservoir to permit full development of the withdrawal limits. The result is that the unobstructed limit is moved vertically to draw in the required flow. Also, the approach geometry to the outlet structure may affect the withdrawal zone through reduction of the lateral extent of withdrawal or the withdrawal angle.

For example, an outlet structure placed at the end of a long, narrow channel will have a greater vertical withdrawal zone as compared to a similar structure that is located on the main body of a lake and is able to withdraw water from a full 180 degrees or greater around it.

For existing projects for which selective withdrawal is being considered, the existing release quality must then be simulated. This may involve the use of a numerical model, such as SELECT, or a one-dimensional model such as WESTEX or CE-QUAL-Rl. For proposed projects, simulation will require a one-dimensional model. In some cases where local topography may influence the withdrawal zone, a physical model of the intake structure and local topography may be necessary to determine impacts. Simulations should then be verified against project conditions.

All of these options and scenarios have been incorporated into the computer program SELECT (Davis et al. 1987). Given a particular water surface elevation, temperature profile and center-line elevation, and the dimensions and withdrawal flows of a port, the withdrawal zone can be predicted. The results predict the upper and lower withdrawal limits as well as a withdrawal velocity profile and the resulting average outflow parameters (i.e., temperature).

If profiles of other conservative parameters (such as dissolved oxygen and algal profiles) are used in combination with the temperature profile, SELECT will also predict the resulting average outflow qualities. It is also possible to adjust the program to account for many differing entrance conditions.

At this point in the design, the location of potential new ports should be determined. The numerical model used above to verify existing conditions is then used to predict release conditions with the new port(s) locations. If multiple ports are necessary, optimization routines are available to determine the optimum number and location for new ports.

By knowing the temperature profile as well as profiles of any other important qualities in front of the outlet structure, and having ports at different elevations throughout the water column, it is possible to withdraw a wide range of qualities provided they are available in the lake. By operating multiple ports, it becomes possible to mix water of different qualities to produce a desired quality.

4.3.1.4 Applications of Selective Withdrawal

The selective withdrawal phenomenon has been of interest to researchers for the past 40 years. As such, there are many reports detailing lakes and reservoirs in which this technique has been tested and used successfully to solve water quality concerns. In addition, several other techniques discussed in this report utilize selective withdrawal for identification or evaluation purposes.

Much work has been done toward describing and predicting how water will behave when withdrawn from a stratified environment. Cariel (1949), Bohan and Grace (1969), Monkmeyer et al.

(1977), Fontane, Labadie, and Loftis (1982), and Smith et al. (1987) have performed research into density-influenced flow and selective withdrawal. These references examine some of the governing equations as well as theoretical, laboratory, and field work to document this phenomenon.

The US Army Engineer Waterways Experiment Station (1986) compiled a reference document on selective withdrawal which presents design considerations for selective withdrawal structures as well as operational experiences and guidance.

Results and descriptions of numerical modeling of the selective withdrawal process are presented by Howington (1989) for operation of a selective withdrawal structure at Lost Creek Reservoir in Oregon. Selective withdrawal was also used in the thermal analysis of Prompton Reservoir (Price and Holland 1989). In this investigation, the number and location of selective withdrawal ports were identified for a proposed modification to raise the pool elevation. Optimization techniques included with the model that was used resulted in a minimum number of ports required to maintain downstream release temperature.

Maynord, Loftis, and Fontane (1978) conducted a study of the proposed Tallahala Creek Lake in Mississippi to determine an optimum design for the intake structure to supply the necessary water quality. Davis et al. (1987) documented the one-dimensional numerical model SELECT and described its operation and limitations.

4.3.1.5 Summary

Selective withdrawal is a technique used to identify the withdrawal zone for a given structure. Using this knowledge, modification of the structure or operation to achieve a given release objective may be possible. Selective withdrawal is used as part of several of the other techniques mentioned in this report. The theory is based on identification of the density-impacted withdrawal pattern. Numerous research and specific reports have been published on selective withdrawal. Most computational methods for selective withdrawal have been included in the computer program SELECT. This program provides predicted release characteristics based on input profiles and operating conditions. Table 4.3.1 presents a summary of this technique.

Table 4.3.1

Summary of Multilevel Selective Withdrawal

Characteristic	Description
Target	To withdraw water from a range of elevations in the reservoir to obtain the highest quality possible.
Mode of action	Release ports at different elevations in the reservoir allow access to a wide range of water quality conditions under stratified conditions.
Effectiveness	Very effective.
Longevity	Years, life of the project.
Negative features	Additional maintenance and operational costs over a simpler structure.
Costs	Expensive.
Applicability to reservoirs	Very applicable to reservoirs with a range of conditions that vary vertically in the reservoir.

4.3.2 HYPOLIMNETIC WITHDRAWAL

4.3.2.1 Description

In many reservoirs and lakes, thermal stratification during summer, combined with an oxygen demand, results in anoxic conditions in the hypolimnion. This anoxic condition allows phosphorus from internal sinks to be released to the hypolimnion and ultimately contribute to the eutrophication of the lake. Selective evacuation of the hypolimnion, termed hypolimnetic withdrawal, can remove excess phosphorus concentrations from the lake and reduce the rate of eutrophication (Figure 4.3.2).

4.3.2.2 Theory

Hypolimnetic withdrawal is a form of selective withdrawal, but with only the release of water from the hypolimnion. Epilimnetic water, which maintains adequate concentrations of dissolved oxygen, is retained in the lake. According to Nurnberg (1987), the major objective is the reduction of anoxic conditions in the hypolimnion which, in turn, will limit the release of phosphorus from the sediments and reduce the cycling of nutrients to the epilimnion. This can be accomplished by changing the location of the withdrawal from the epilimnion to the hypolimnion. On an annual basis, the volume of water released remains unchanged but the

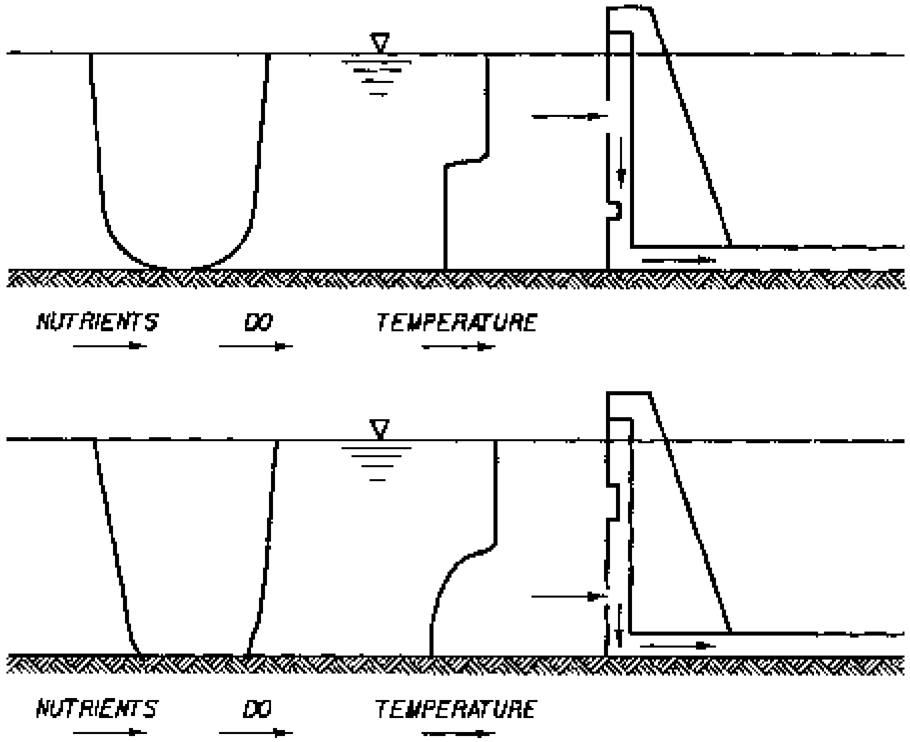


Figure 4.3.2 Nutrient reduction through hypolimnetic withdrawal

thermal stability may be reduced, depending upon the lake's morphology and inflow water quality characteristics.

This technique does not necessarily lead to destratification in the traditional sense, since it involves removing the hypolimnetic water and not mixing it with the epilimnetic waters. Traditional destratification would enhance nutrient transport from the hypolimnion to the epilimnion and accelerate eutrophication. A possible consequence of this technique is the release of poor-quality (low dissolved oxygen and high dissolved solid) hypolimnetic water downstream.

4.3.2.3 Evaluation Methodology

General design guidance was given by Nurnberg (1987) for hypolimnetic withdrawal in lakes. The design approach given below is that of Nurnberg with additional criteria for large reservoirs.

The design of a hypolimnetic withdrawal system should begin with a clear definition of the existing stratification structure and nutrient budget of the lake. The analysis of the thermal structure requires temperature and DO profiles during the stratified period. These should be at sufficient intervals to describe the stratification process, the development of anoxic conditions in the hypolimnion, and the subsequent breakup of stratification in the fall. Concurrent nutrient profile data are also required to determine the contribution of nutrients from the hypolimnion.

The effects on the nutrient budget of releasing a significant portion of the hypolimnion should be estimated. Several numerical techniques and simulation models are available which may assist in this process. If the numerical studies predict little or no change in nutrients, then further analysis is unnecessary and alternative techniques must be investigated. If, however, there is a reduction of epilimnetic nutrient concentration without significantly impacting total lake volume, then a hypolimnetic withdrawal analysis should be conducted.

If a hypolimnetic outlet exists in the release structure, the numerical water quality models can be used to determine the maximum release flow and yet minimize the release of epilimnetic water. If a new hypolimnetic port is needed, its elevation, as well as the flow rate, can be determined. This analysis should be conducted over the entire stratification season and will likely require the use of a one- or two-dimensional water quality model.

4.3.2.4 Review of Applications

This technique has been used on a number of lakes that are relatively small, with the largest being 78,570 acre-ft (97 million cubic meters) (Nurnberg, Hartley, and Davis 1987). Although smaller reservoirs may be suited for this technique, application to large reservoirs has not been attempted. Results of hypolimnetic withdrawal at a number of lakes indicated that hypolimnetic withdrawal is an effective technique for the reduction of nutrients in the epilimnion; however, impacts on anoxia in the hypolimnion are inconclusive. A summary of hypolimnetic withdrawal projects with relevant hydrologic and morphometric information is given in Nurnberg (1987).

4.3.2.5 Summary

Hypolimnetic withdrawal is an application of selective withdrawal designed to remove anoxic hypolimnetic water from a reservoir. The anoxic conditions allow nutrients to be released into the epilimnion, contributing to eutrophic conditions. By releasing hypolimnetic water instead of epilimnetic water, a reduction in the internal nutrient cycling can be achieved. This technique uses the stratification of the reservoir to maximize releases from the hypolimnion to reduce its volume. Although this technique may be applicable to smaller reservoirs, it has not been implemented at large reservoirs. Pertinent information is summarized in Table 4.3.2.

Table 4.3.2

Summary of Hypolimnetic Withdrawal

Characteristic	Description
Targets	Reservoir nutrient cycling and anoxic conditions (and associated concerns) in the hypolimnion.
Mode of action	Removal of hypolimnetic waters through bottom withdrawal.
Effectiveness	Effective on small lakes with strong stratification, dependent on lake volume.
Longevity	Seasonal.
Negative features	Removal of hypolimnetic zone will reduce or eliminate habitat for cold-water fisheries. Downstream release water will typically be of poor quality.
Costs	Minimal.
Applicability to reservoirs	Applicable to reservoirs with withdrawal facilities whose intakes are located in the hypolimnetic region of the reservoir.

4.3.3 UNDERWATER SKIMMING WEIR OR SUBMERGED CURTAINS

4.3.3.1 Problem Addressed

Reservoirs that exhibit strong thermal stratification with a cool hypolimnion throughout the summer months may be able to maintain a cool-water fishery in the hypolimnion or in the river downstream. If the project operates a bottom withdrawal structure, such as a hydropower project, the cool hypolimnetic water can be deplete during the summer months. Therefore, maintenance of a cool-water fishery is difficult, if not impossible. A temperature control curtain or submerged skimming weir (underwater dam) can retain the cool water by preventing or limiting its withdrawal and release downstream (Figure 4.3.3). If a curtain is used, it can be lowered to allowing withdrawal of “stored” cool water in the late summer, thereby maintaining water quality for a cool water fishery. However, underwater dams may hinder reservoir drawdown, act as a sediment trap, and impede fish migration.

4.3.3.2 Theory

In the normal operation of a reservoir with a low-level intake, during the summer, the hypolimnion is gradually withdrawn and released downstream. Thus, the cooler hypolimnetic water is gradually replaced with warmer water from the mixed surface layer. Although thermal stratification may be retained, the degree of stratification is weakened such that the surface-tobottom temperature difference is continually being reduced. This reduces the available habitat for in-reservoir cool-water fish and may threaten their survival. If the cool-water fishery is in the downstream river, the loss of cool water for release can be likewise disastrous.

A temperature control curtain or a submerged weir can limit the withdrawal of hypolimnetic water preserving the cool water for in-reservoir fishery or for later release to maintain cool temperatures in the downstream river. A cool-water "sanctuary" can also be created in an embayment of the reservoir, but fish usage may be limited. The temperature control weir or curtain modifies the withdrawal zone of the structure's outlet to minimize release of cool water by skimming release water from the epilimnion. If there is cool water fishery in the downstream river, then temperature control is aimed at maintaining cool water withdrawal from the reservoir. A temperature control weir, installed in front of the outlet, can limit withdrawal of epilimnetic water by skimming cool water from the bottom of the reservoir.

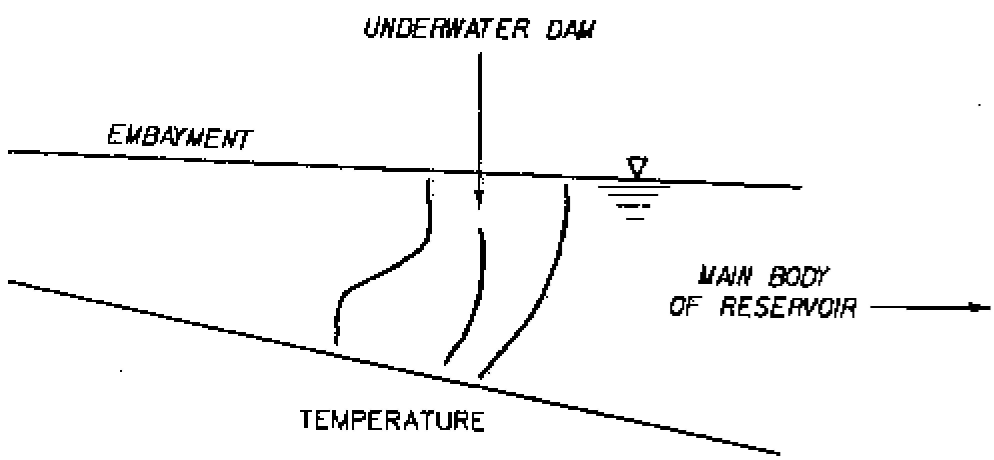


Figure 4.3.3 Schematic of a submerged dam

The design procedure is similar to the design of any withdrawal structure:

a. determine objectives of the structure. The temperature control device be designed to skim surface water from the reservoir and retain cool water in the reservoir or regulate the release of cool water to meet temperature objectives in the downstream river. The latter requires the temperature curtain or weir have operational capabilities to adjust the withdrawal zone with changes in reservoir stratification.

b. determine the appropriate type and structure design. Select a curtain or earthen/rock fill weir as the structure. Although rock or stone, sheet pile, and fabric liners are feasible construction materials, an impermeable fabric material seem to be the least expensive. This however, requires the designer to consider the substantial force that the cool dense water will exert on the curtain with in place. Use the numerical model SELECT (Davis, et al 1987) or a numerical water quality model to estimate the required elevation and dimensions for the outlet device. The thermal stratification history of the reservoir should be reviewed to determine the lowest elevation of the thermocline which will be the crest elevation for the underwater dam.

c. Vermeyen (1997) provides guidance on the design, fabrication, installation, and operation of temperature control curtains based on Bureau of Reclamation experience at 4 reservoirs. Bohac (1989) also describes a design procedure used by TVA for a submerged curtain constructed at Cherokee Reservoir. Bohac, Baker, and Shane (1986) present site-specific design information for the Cherokee curtain.

4.3.3.3 Applications

This relatively new technique has been implemented at several projects in the US Bureau of Reclamation (USBR) and the Tennessee Valley Authority (TVA). The technique has been successful in establishing a cool water refuge for fish (Bohac 1989) in Cherokee Reservoir. The Bureau of Reclamation has successfully employed curtains to control release temperatures at Lewiston and Whiskeytown Reservoirs in California and the Bureau has designed a control curtain for Shasta Dam.

4.3.3.4 Summary

A submerged weir or temperature control curtain is an outlet structure designed to skim warm epilimnetic water for release to meet warm temperature requirements downstream or retain cool hypolimnetic water in the reservoir. In some instance, curtains have been used to skim cool water from the hypolimnion of reservoir to meet cool-water objectives in the river. The technique was also used to create a cool-water habitat in a reservoir embayment. Design and construction guidelines are presented by Vermeyen (1997). Information concerning this technique is summarized in Table 4.3.3.

4.3.4 SUBMERGED SKIMMING WEIR

4.3.4.1 Problem Addressed

The thermal stratification that occurs at most reservoirs can create problems with release water quality. If the intake structure withdraws water from the hypolimnion, the release may be low in dissolved oxygen (DO) or contain high concentrations of undesirable trace constituents. In t is case, withdrawal of water primarily from the epilimnion may be desirable. If the project must release high discharges, as for a hydropower project, discharge through a port (single port or horizontal line of

Table 4.3.3

Summary of Temperature Control Weirs and Curtains

Characteristic	Description
Targets	To modify withdrawal characteristics to skim surface water or skim cool hypolimnetic water for release. To create a cool-water refuge for cold-water fisheries.
Mode of action	An underwater weir or curtain is constructed to skim either surface or hypolimnetic water, depending upon operational objectives.
Effectiveness	Depends on stratification.
Longevity	Years.
Negative features	May limit the lateral exchange of waters between the main lake and the cold-water refuge.
	May act as a sediment trap.
	Limits the migration patterns of certain species of fish.
Costs	Moderate to high (site and construction specific).
Applicability to reservoirs	Very applicable, but feasibility and effectiveness are site specific.

ports) will result in a large withdrawal zone that usually extends surface to bottom. To increase the amount of epilimnetic water in large releases, a submerged skimming weir can be designed and installed. This structure is designed to extend to th6 top of the reservoir thermocline and prevent the withdrawal of the hypolimnion.

This technique should be used only at projects with minimal lake surface elevation fluctuation during the period of hypolimnetic anoxia, since the effectiveness of the weir will diminish as a function of the drawdown on the reservoir. In addition, a weir can strengthen the thermocline in the main portion of the lake. This can affect a number of related concerns, such as anoxic conditions in the hypolimnion and nutrient cycling. A weir may also act as a sediment trap, which may or may not be beneficial.

4.3.4.2 Theory

The theory of operation of a submerged skimming weir is based on the modification of the withdrawal limits by the weir crest. Using the epilimnetic water quality as the desired release quality, operation of a weir can be simulated, and the resulting quality predicted for a given weir crest elevation, thermal stratification, and discharge. A numerical model can be used to simulate various stratification patterns and discharges and to determine the optimum crest elevation for the weir (Figure 4.3.4). The effects of various meteorological and hydrological events on the reservoir operating with a submerged weir can then also be simulated.

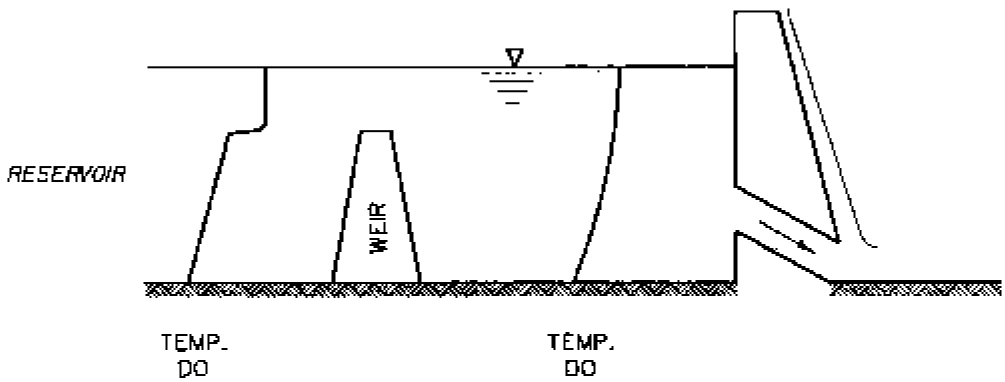


Figure 4.3.4 Schematic of submerged skimming weir

4.3.4.3 Design Methodology

Design of a submerged skimming weir should begin on ly after clearly defining the existing release water quality problems. The design methodology is based on installation of submerged skimming weirs at existing Corps of Engineers' (CE) projects.

To determine this, a numerical model simulation can be used to predict the reservoir quality and stratification under a variety of meteorological, hydrological, and operational conditions. If the project is already operational, simple examination of release quality and in-reservoir quality profiles will indicate if a problem exists. If the problem involves release of hypolimnetic water, such as low DO during stratified periods, a submerged skimming weir may be a viable alternative to minimize hypolimnetic releases.

The numerical model simulations described above, conducted to determine the viability of a submerged weir, can also be used to determine the crest elevation of the weir. The crest is usually set at an elevation equal to or higher than the highest thermocline elevation observed during the summer. If the thermocline elevation is above the weir crest elevation, withdrawal of hypolimnetic water can occur, thereby diminishing weir effectiveness.

Once the crest elevation has been selected, the crest length must be determined. Although the shorter weir lengths are less expensive to construct, their effectiveness can also be reduced. For example, a weir crest with the same length as the intake port may develop a withdrawal zone very similar to the port because of the withdrawal of hypolimnetic water over the weir. Therefore, some reasonable length must be determined to begin design. Using an initial length, the numerical simulations are repeated as in the initial evaluation of a submerged weir. Examination of the results for the depth of the withdrawal zone and the resulting release water quality will determine the optimum weir length.

Some discretion in the length of the weir must be used, as well as site-specific knowledge of the area in front of the intake. In most applications, the weir crest should tie into the same elevation on either bank in front of the structure to prevent the withdrawal of hypolimnetic water from around the weir.

4.3.4.4 Applications of Submerged Skimming Weirs

Submerged skimming weirs have been used at several CE projects to prevent the release of low-DO hypolimnetic water. Linder (1986) described the design and evaluation process for submerged skimming weirs at two hydropower projects in the Kansas City District. These projects, Stockton and Harry S. Truman Dams, were predicted to have low dissolved oxygen in the hydropower releases. During construction, provisions were made to construct weirs as part of the project. Evaluation of this technique at these projects indicates that the release quality is improved, with some minor elevation of release temperature in the fall and some cooling in the spring.

Another concern that developed after construction of the projects was that, during long periods of nongeneration, stratification of the area between the weir and the dam can occur. Upon start-up of generation, some low-DO water is released downstream. This problem has also been documented at Clarence Cannon Dam in the St. Louis District (Wilhelms and Furdek 1987). However, the submerged weir at Clarence Cannon functions as designed during generation and maintains adequate levels of dissolved oxygen in the hydropower releases.

An evaluation of a skimming weir for temperature control at Meramec Park Dam, a project planned for the Meramec River in Missouri, indicated that a weir would be effective for controlling cool-water releases during the stratified period (Bohan 1970). However, the project was deauthorized and never constructed. An evaluation of a submerged skimming weir at Richard t. Russell was performed to assist in release water quality enhancement (Smith et al. 1981). Results indicated that the weir, which was proposed as a modification of an existing cofferdam, would interfere with the hydrodynamics of proposed pumpback operations and therefore was not recommended for construction.

The US Bureau of Reclamation (USBR) has also considered the use of submerged weirs to enhance release water quality. Shasta Dam, located on the Sacramento River, was evaluated for retrofit of a plastic curtain to serve as a submerged weir and allow the release of warmer surface water. It was also to be designed such that it could be raised to permit the release of bottom water (CH2M Hill 1977). Although evaluation of this technique indicated that release quality could be met, some concern over potential blockage of hydropower intakes if the curtain broke free from moorings has delayed implementation of this technique.

4.3.4.5 Summary

The submerged skimming weir is a technique that will modify,-the withdrawal zone of the release structure, thereby modifying the release quality during the stratified periods. This technique has been used at large CE projects for maintaining release quality at peaking hydropower projects that maintain a relatively stable pool elevation. A design approach using numerical models to simulate the effects of hydrological and meteorological conditions on the reservoir was identified. Several projects for which weirs have been evaluated and/or installed were discussed. Summary information on this technique is presented in Table 4.3.4.

Table 4.3.4

Summary of Submerged Skimming Weir

Characteristic	Description
Target	Improvement of the release water quality by restricting the withdrawal to the epilimnion of the reservoir.
Mode of action	A submerged weir in the vicinity of the intake is used to block the withdrawal of hypolimnetic water and allow the release of epilimnetic water.
Effectiveness	Effective at given flow rates.
Longevity	Years.
Negative features	May strengthen the thermocline in the main part of the lake.
	Limits the transport of sediments thr6ugh the project.
	May increase the anoxic conditions (and related concerns) in the hypolimnion.
Costs	Expensive (construction).
Applicability to reservoirs	Very applicable, even to reservoirs with selective withdrawal structures.

4.3.5 References

Bohac, C. E. 1989 (Apr). "Underwater Dam and Embayment Aeration for Striped Bass Refuge," *Journal of Environmental Engineering*, ASCE, Vol 115, No. 2, pp 428-446.

Bohac, C. E., Baker, J. M., and Shane, R. M. 1986. "Cherokee Reservoir Striped Bass Refuge Demonstration Project Design, Construction, and First-Year Operation," TVA/ONRED/AWR-86/27, Division of Air and Water Resources, Tennessee Valley Authority, Chattanooga, TN.

Bohan, J. P. 1970. "Water Temperature Control Weir for Meramec Park Dam, Meramec River, Missouri," Technical Report H-70-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Bohan, J. P., and Grace, J. L., Jr. 1969. "Mechanics of Flow from Stratified Reservoirs with Interest of Water Quality; Hydraulic Laboratory Investigation," Technical Report H-73-4, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

CH2M Hill. 1977 (May). "Conceptual Design, Shasta Dam Emergency Temperature Control," Redding, CA.

Davis, J. E., Holland, J. P., Schneider, M. L., and Wilhelms, S. C. 1987. "SELECT: A Numerical, One-Dimensional Model for Selective Withdrawal," Instruction Report E-87-2, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Fontane, D. C., Labadie, J. W., and Loftis, B. 1982. "Optimal Control of Reservoir Discharge Quality Through Selective Withdrawal; Hydraulic Laboratory Investigation," Technical Report E-82-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Gariel, P. 1949. "Experimental Research on the Flow of Superposed Layers of Fluids of Different Densities," *LaHouille Blanche*, Vol 4, No. 1.

Howington, S. E. 1989. "Intake Structure Operation Study, Lost Creek Dam, Oregon,' Technical Report HL-89-13, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Linder, W. M. 1986 (Feb). "Field Experience with Skimming Weirs in the Kansas City District," *Proceedings of a Seminar on Water Quality R&D: Successful Bridging Between Theory and Applications*, US Army Corps of Engineers Committee on Water Quality, Washington, DC, pp 153-171.

Maynord, S. T., Loftis, B., and Fontane, D. G. 1978. "Temperature Analysis and Selective-Withdrawal Design Study, Tallahala Creek Lake, Mississippi; Mathematical Model Investigation," Technical Report H-78-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Monkmeyer, P. L., Hoopes, J. A., Ho, J. C., and Clark, G. R. 1977. "Selective Withdrawal and Heated Water Discharge: Influence on the Water Quality of Lakes and Reservoirs; Part I: Selective Withdrawal," Technical Report WIS WRC 77-03, Water Resources Center, University of Wisconsin, Madison, WI.

Nurnberg, B. K. 1987 (Oct). "Hypolimnetic Withdrawal as Lake Restoration Technique," *Journal of Environmental Engineering*, Vol 113, No. 5, pp 1006 1017.

Nurnberg, B. K., Hartley, R., and Davis, E. 1987. "Hypolimnetic Withdrawal in Two North American Lakes with Anoxic Phosphorus Release from the Sediment," *Water Resources*, Vol 21, No. 8, pp 923-928.

Price, R. E., and Holland, J. P. 1989. "Thermal Analysis of Prompton Reservoir Modification, Pennsylvania," Technical Report HL-89-20, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Smith, D. R., Holland, J. P., Loftis, B., and Tate, C. H., Jr. 1981. "Evaluation of In-Reservoir Cofferdam on Richard B. Russell Reservoir and Hydropower Releases," Technical Report HL-81-12, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Smith, D. R., Wilhelms, S. C., Holland, J. P., Dortch, M. S., and Davis, J. E. 1987. "Improved Description of Selective Withdrawal Through Point Sinks," Technical Report E-87-2, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

US Army Engineer Waterways Experiment Station. 1986. "Proceedings: CE Workshop on Design and Operation of Selective Withdrawal Intake Structures," Miscellaneous Paper HL-86-3, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Vermeyen, Tracy. 1997. "Use of Temperature Control Curtains to Control Reservoir Release Water Temperatures," Report No. R-97-09, US Department of the Interior, Bureau of Reclamation, Water Resources Research Laboratory,

Wilhelms, S. C. 1986 (May). "Selective Withdrawal: Basic Concepts," in "Proceedings: CE Workshop on Design and Operation of Selective Withdrawal Intake Structures," Miscellaneous Paper HL-86-3, US Army Engineer Waterways Experiment Station, Vicksburg, MS, pp 13-18.

Wilhelms, S. C., and Furdek, T. J. 1987. "Local Destratification System for Mark Twain Lake," *Proceedings: CE Workshop on Reservoir Releases*,-Miscellaneous Paper E-87-3, US Army Engineer Waterways Experiment Station, Vicksburg, MS, pp 137-141.

4.4 IN-RESERVOIR TECHNIQUES - OPERATIONAL TECHNIQUES

4.4.1 OPERATIONAL CURVE CHANGE

4.4.1.1 Description

Many reservoir projects operate under some type of water control plan that relies on seasonal change of reservoir elevations. This sequence of reservoir elevation change is usually termed a guide (or rule) curve. Depending on the project authorization, the guide curve may permit rather large pool-level fluctuations on an annual basis. For example, operation of a flood control project usually involves drawing down the reservoir level during the fall and winter and filling during the spring and early summer to a stable pool through the summer and early fall.

In some cases where the summer pool elevation is maintained at a relatively stable elevation, water quality may be a concern. Water quality concerns in the reservoir which result from poor quality inflows may be averted by pooling up later in the year. In another approach, the retention time of the reservoir may be modified by adjusting the guide curve. Constituents of water quality that may be affected by this technique include inflow with undesirable qualities, nutrient loading of the reservoir (and, in turn, the effects on algal growth and fisheries), turbidity, and sedimentation.

For example, inflows with a high sediment load may be delayed in the upper reaches of a reservoir and allowed to settle out before reaching the outlet works. Therefore, some modification of the guide curve to reduce or increase the residence time may enhance the water quality of the reservoir.

Modification of the guide curve is a technique that relies on changing the hydraulic residence time of inflows in the pool to control poor quality inflows. This can be accomplished by maintaining a small pool through most of the flood control season and allowing undesirable inflows entering the reservoir to be flushed through. The reservoir is then allowed to fill to summer pool elevation later in the season, when the inflow water quality is typically better. If inflow quality is governed more by discharge than season, maintaining a larger pool with a greater retention time may allow suspended material to settle in the headwaters, and thereby improve the quality of the main portion of the reservoir.

Another variation of this technique relies on changing the minimum pool elevation of the reservoir and allowing additional storage for water quality maintenance or conservation of a quality resource, such as cool water for downstream release. Adjusting the guide curve may also have a significant benefit to the wildlife of the reservoir by creating additional habitat at critical times or controlling aquatic plant growth. An example of a modification of a guide curve is shown in Figure 4.4.1.

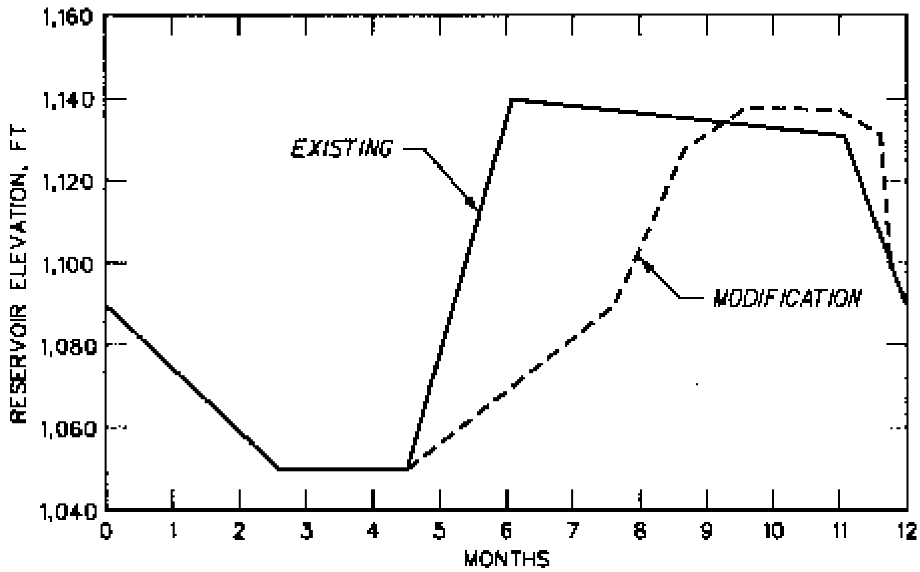


Figure 4.4.1 Example of guide curve modification

4.4.1.2 Design Criteria

The implementation of a proposed change to the water control plan or guide curve should occur only after the water quality objectives of a proposed change are stated. This will involve identification of existing water quality parameters of concern and the desired goals. Since no general procedure for the evaluation of a proposed change in the water control plan for a reservoir project has been published, a method for evaluating a guide curve change is given below. The specific objective of the guide curve change must be determined, such as reduced turbidity in the reservoir. This will involve determination of existing water quality parameter concentrations, such as turbidity levels, in the reservoir relative to the existing guide curve. Next, the flexibility for change of the guide curve or the upper and lower limits of the curve must be examined to determine the amount of change available. This must be considered within the context of the authorized project purposes. The impacts of the proposed change of the guide curve on the water quality objective must then be determined. This may require a variety of assessment techniques ranging from simple computations of hydraulic residence time to two-dimensional numerical models. The level of computational sophistication required is predicated on the water quality objective and degree of change of the guide curve. Relatively small changes in the guide curve, which may have minimal impacts on the water quality objective or other project purpose, may be evaluated with residence time and stability computations. Large changes, which may expand the size and volume of the pool, may require a numerical model investigation to evaluate the impacts of the change.

The impacts of the proposed change on the release water quality must be evaluated. If a significant change in the guide curve is proposed, impacts to the release water quality may be possible. For example, increased residence time to allow settling of suspended solids may enhance thermal stratification and subsequent depletion of oxygen in the hypolimnion. Withdrawing water from the lower elevations of the reservoir could result in poor downstream water quality and possibly offset any benefits achieved by the guide curve change. The delay in reservoir filling may also affect other areas. These include recreation and access to fishing and swimming areas, as well as docks and marinas; possible erosion of the exposed lake bottom due to wave action and the lack of a stabilizing plant cover; the aesthetics of the exposed shoreline; management of aquatic plants; impacts on fish and wildlife habitat; and effects on access to wetlands and spawning areas.

4.4.1.3 Implementation

Applications of guide curve changes or modifications to guide curves have not been widely reported in the literature. However, the control of hydraulic residence time has been reported from laboratory studies to be a feasible technique in certain instances to enhance reservoir water quality (Schiebe and Dendy 1978). In addition, this technique is a recognized method to enhance water quality at Corps of Engineers' (CE) projects (HQUSACE 1987b).

A general evaluation technique for water quality forecasting for short and long-term impacts of a guide curve modification may be found in "Management of Water Control Systems" (HQUSACE 1987a). This technique relies on the expertise of the water control manager in deciding on which approach to take with a particular guide curve modification. The use of computer models to simulate short- and long-term impacts is recommended.

A number of techniques that use numerical models have been developed to evaluate guide curves for water quantity or volume perspectives (Newman and Loucks 1975). Recent investigations have added water quality objectives to the development of guide curves (Hogan 1986). This technique has combined numerical optimization methods with a numerical model that simulates the physical processes in the reservoir to optimize the real-time operation of a reservoir.

This technique has been used to improve water quality during the initial filling sequence of new reservoirs. Many new reservoirs, especially large ones, are filled in stages over a number of years to minimize the inundation of new sediment and the oxygen demand of this material. Van Pagee et al. (1982) identified a delayed filling period as one alternative to minimize the impacts of filling of a new reservoir in the tropics region in Suriname. A similar technique is discussed in Engineer Manual 1110-2-1201 (HQUSACE 1987b).

4.4.1.4 Summary

The modification of guide or rule curves to water storage in reservoirs can be used as an operational technique to enhance water quality. By modifying the hydraulic residence time of the reservoir, inflows can be routed quickly through the reservoir or retained. This allows the reservoir to retain high quality water for later release or retain poor quality water for treatment by in-reservoir processes to modify its quality.

A numerical water quality model can be used to evaluate changes in the operational guide or rule curve. Although few examples of guide curve changes have been published, some referenced examples indicate positive benefits to this technique. Table 4.4.1 summarizes the guide curve change technique.

Table 4.4.1

Summary of Guide Curve Change

Characteristic	Description
Targets	Nutrient and sediment loading, turbidity, and poor water quality inflows.
Mode of action	Adjust scheduled reservoir pool-up to change residence time and minimize the effects of poor water quality inflows.
Effectiveness	Highly effective on small- to medium-sized reservoirs.
Longevity	Seasonal
	Possible loss of fish and wildlife habitat and/or access to wetlands or spawning areas during critical hydrologic events.
Negative features	Access to recreation areas may be limited.
	Aesthetics, shoreline erosion, and aquatic weed control affected.
	Risk as to being able to fill the reservoir to summer levels in the season
Costs	Minimal
Applicability	Small- to medium-sized reservoirs, where volume change will significantly affect the residence time.

4.4.2 INFLOW ROUTING

4.4.2.1 Description

Inflows to reservoirs can create problems with reservoir water quality. If the inflow quality is poor, containing high concentrations of nutrients, suspended solids, or other undesirable constituents, poor reservoir water quality may result. Depending on the volume of the inflow and the retention time of the reservoir, the inflow constituents may settle and be trapped in the reservoir. This could contribute to eutrophication processes as well as filling of the reservoir with sediment. If inflow quality is identified as a concern, it may be possible to route the inflow through the reservoir for release downstream, without significantly impacting the quality of the reservoir, or hold the inflow in the project for treatment while releasing unaffected water downstream.

Routing of inflows is a technique based on the water control operation or plan for the reservoir. Undesirable inflows are identified and routed through the reservoir to minimize impacts to the existing reservoir water quality, similar to routing of flood flows through a basin. Upon identification of an undesirable inflow, the route the inflow will take in the reservoir as well as the volume of the inflow must be predicted. This can be accomplished through a combination of upstream flow gauging and density flow calculations similar to the techniques derived by Akiyama and Stefan (1985).

Since the inflow will seek its layer of neutral density in a density stratified reservoir, a density current will develop and proceed through the reservoir on the surface or at some elevation, depending on the stratification. Using the existing release structure and operation within the existing water control plan, the project is operated to move the undesirable water through the pool as quickly as possible (Figure 4.4.2).

Selective withdrawal capability greatly enhances this technique if the inflow occurs at some intermediate depth. However, depending on the equilibrium elevation of the inflow, it may be possible to use other outlet works' (i.e., spillways or sluiceway) to release the inflow. Because no structural modification or addition is involved in this technique, costs are associated only with evaluation and change of operation.

4.4.2.2 Design Criteria

The methodology for this technique involves definition of the goals to be achieved relative to the reservoir water quality. This will usually involve determination of the reservoir water quality that exists prior to some inflow event, predicted quality of the inflow, and its impacts on the reservoir. A general procedure for routing of undesirable inflows is given below.

The specific water quality of the reservoir and inflow as well as the resulting reservoir quality must be determined or predicted. This will usually involve inflow temperature, turbidity,

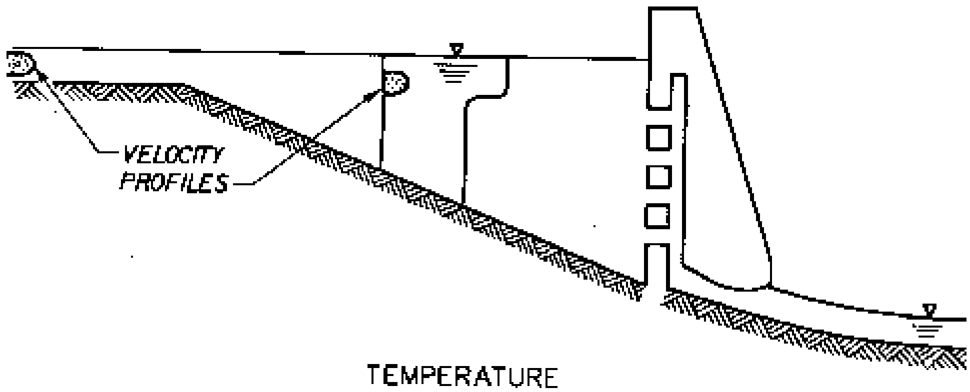


Figure 4.4.2 Routing of undesirable inflows

suspended solids, nutrients or other constituents, and corresponding concentrations in the reservoir.

Using the data collected, as well as a knowledge of the flow patterns through the reservoir, the location and elevation of the inflow in the pool must be predicted. Akiyama and Stefan (1985) derive the equations necessary to predict the routing of turbidity currents (a form of density current) through a reservoir. It may be possible to incorporate these equations into a numerical model to help the reservoir manager make accurate, real-time predictions of the flow events. However, mixing of the inflow with reservoir water, as well as dispersion in the reservoir, may be difficult to predict, even with a numerical model.

Using the predicted location of the undesirable inflow in the reservoir, the water control plan should be evaluated to determine appropriate operations to minimize impacts to the reservoir. This may involve varying the release flow rate or changing the selective withdrawal port elevation to withdraw the inflowing water from a specific elevation more efficiently. Since retention of the undesirable inflow in the pool is likely detrimental to the water quality of the reservoir, release of these flows should be maximized.

4.4.2.3 Implementation

Although application of this technique may be standard procedure for most water quality managers, published accounts of inflow routing are limited.

Nix (1981) reported that the adjective transport of organic matter resulting from an inflow event contributed to the oxygen demand of a reservoir metalimnion during the stratified periods. He indicated that residence time of this organic matter could be minimized by increasing the discharge rate of the project through the stratification season.

Iwasa and Matsuo (1981) discussed the problem of turbid inflows and the impacts to the reservoir water quality. When the reservoir lacks thermal stratification, turbid inflows mix the entire reservoir and create a linear turbidity profile from surface to the bottom. When thermal stratification is present, the turbid inflow remains near the thermocline and does not easily settle into the hypolimnion. The authors indicated that mathematical modeling techniques could be used to control reservoir turbidity by withdrawing the turbid layer.

In some reservoir systems, inflows occurring in the absence of thermal stratification may be routed through the reservoir without mixing the entire reservoir. Although published accounts of this phenomenon are unavailable, this has been reported through personal communications.[1]
Murota and Michioku (1986) described a selective withdrawal technique for reservoirs that have a three-layer stratification. Their discussion also indicated the usefulness of removing layers in the reservoir that are highly turbid and may contribute to sedimentation in the reservoir.

4.4.2.4 Summary

Inflows to reservoirs can create water quality problems, if they contain significant concentrations of suspended solids, turbidity, or other undesirable constituents. By quickly routing these undesirable constituents through the reservoir, the impacts can be minimized.

A simple concept is presented to evaluate the effectiveness of routing of inflows for water quality purposes. Although few references are available for this technique, the approach may be useful for some projects. A summary of information concerning this technique is presented in Table 4.4.2.

4.4.3 SUPPLEMENTAL RELEASES FOR WATER QUALITY

4.4.3.1 Description

Many multipurpose projects incorporate several types of release structures, designed for flood control, hydropower, water quality control, and navigation. In most cases, water control plans do not call for simultaneous operation of the various release structures during the stratification season. For example, flood control and water quality releases will not generally be made at the same time. It may be possible to simultaneously operate two or more release structures and thereby create a selective withdrawal configuration, provided the intake center-line elevations for the various structures are not the same.

Thus, selective withdrawal techniques may be used to enhance reservoir release water quality by reducing the stability of stratification through hypolimnetic withdrawal, enhancing mixing conditions in

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[1] Pete Juhle, Headquarters, US Army Corps of Engineers, Washington, DC.

Table 4.4.2

Summary of Inflow Routing

Characteristic	Description
Targets	Inflows with undesirable water quality constituents.
Mode of action	Selective withdrawal of the inflow
Effectiveness	Dependent on lake size, distance of inflow to outlet works, and strength of density difference between inflow and reservoir.
Longevity	Days to months
Negative features	Technique less effective without selective withdrawal structure. Requires density difference between the inflow and the reservoir (the greater the better). May pass the quality problem on to the river downstream or a downstream reservoir.
Costs	Minimal to none (operating changes and perhaps lost hydropower generation revenue).
Applicability to reservoirs	Works best with reservoirs with selective withdrawal; however, the use of other outlet works is possible

the reservoir, withdrawing undesirable water, or diluting poor-quality release flow with higher quality water. For example, during the stratification season, the quality of hypolimnetic release water may degrade because of significant low dissolved oxygen or concentrations of iron, manganese, and hydrogen sulfide. Releases from the epilimnion through a spillway gate (selective withdrawal) will dilute the poor quality release. Reaeration of releases through a flood control gate may significantly increase oxygen for dilution with the poor quality water.

4.4.3.2 Design Criteria

This operational technique has been used at a number of projects to enhance the release quality during the stratification season. However, the evaluation of the technique has been based on tests in the field. A general procedure for evaluating this technique for the potential to improve-reservoir or release water quality is given below.

Identify the specific water quality concern and objectives. For example, the water quality parameter, such as dissolved oxygen (DO), must be quantified in the reservoir and the release. The existing release characteristics must be determined, including discharge volume and withdrawal characteristics in the pool.

Determine the various release options for a given project. This will involve examination of release options that typically are not made, such as low-flow (minimum) releases from a flood control gate. The improvement of release quality with a modification of the release schedule to include releases from this optional structure must then be evaluated (Figure 4.4.3). In the case of selective withdrawal, the SELECT model (Davis et al. 1987) can be used to predict release quality for most high-head release structure options. If reaeration through the structure is a desired outcome, computational methods to predict the amount of reaeration through a gated conduit, which have been developed by Wilhelms and Smith (1981), may be used. Once the release options have been identified, determine the volume of water required to achieve the stated water quality goal. In most cases, this can be determined using a mass balance approach, where the existing release quantity is diluted with the water quality release.

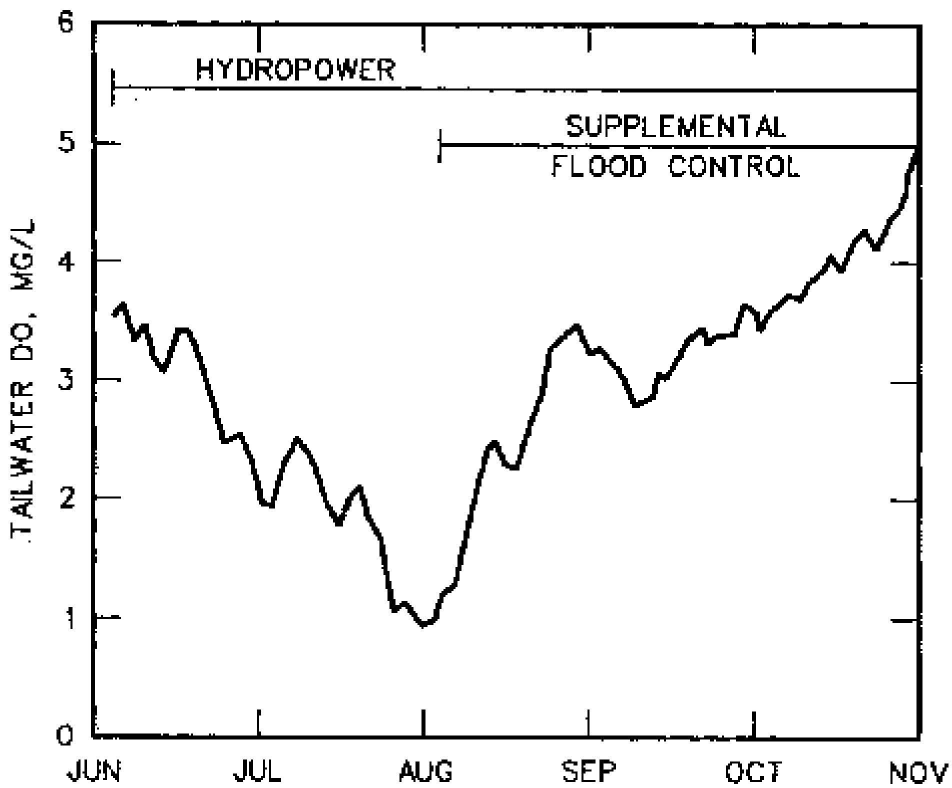


Figure 4.4.3 Example of improved dissolved oxygen with supplemental releases

This procedure has been implemented in a real time frame at several hydropower projects, using a downstream monitoring scheme to identify the need for water quality releases. Once real-time monitoring identifies the need, the water quality release is made.

4.4.3.3 Implementation

This technique has been applied at several projects to improve downstream DO, usually in conjunction with a hydropower operation. However, it can be used with any project in which the water quality concern is a result of stratification in the reservoir, such as release temperature or pH.

At Tims Ford Project, a multipurpose hydropower project operated by the Tennessee Valley Authority (TVA), the need to establish a minimum release from the project during nongeneration periods to maintain the downstream fisheries resource was identified (Goranflo and Adams 1987). The study indicated that 80 cfs should be released. Alternatives that were evaluated included an option to sluice water through the flood control structure. This would provide the required flow and increase the release DO oxygen without requiring a structural modification to the project. A similar evaluation was conducted at the Norris Project with similar conclusions.

Walter F. George Lock and Dam on the Chattahoochee River near Fort Gaines, GA, has experienced problems with low DO in the tailwaters during nongeneration periods in the late summer and early fall. An operational technique that relies on downstream monitoring to identify the onset of low DO conditions is used to enhance the downstream water quality. With the identification of low DO conditions downstream, spillway gates are opened in an ordered manner until downstream DO levels stabilize (Findley and Day 1986). A similar operation that relies on downstream monitoring to identify the need for flood control gate releases has been implemented at Lake Texoma (Price 1990).

Table Rock Dam, located on the White River in Missouri, experiences difficulty meeting DO criteria in the release during the stratification season. Among the alternatives investigated to correct this situation was the use of supplemental releases from the hypolimnion through Howell-Bunger valves. This release would be aerated by the aspiration process associated with these types of valves. The costs associated with loss of generation capacity due to the releases was significantly greater than other alternatives for this project (US Army Engineer District, Little Rock 1985).

4.4.3.4 Summary

The water quality released from a multi-purpose reservoir may be improved by modifying the operation of the various release structures at the project. Specific water quality concerns and enhancement objectives must be identified. For example, release DO may be low and require improvement to some minimal level. Operation of an alternative outlet may permit the release of high-DO water with epilimnetic water or by reaeration for mixing with the poor quality release. Simple techniques are available to estimate the required operational levels to obtain specific goals. Several example applications of this technique were discussed. Table 4.4.3 summarizes this technique.

Table 4.4.3

Summary of Supplemental Releases For Water Quality

Characteristics	Description
Targets	Water quality constituents such as dissolved oxygen (along with related concerns), turbidity, temperature, etc.
Mode of action	Use of release structures, regardless of original purpose, to improve water quality in the reservoir and downstream releases.
Effectiveness	Dependent upon location and elevation of intake of release structures.
Longevity	Days to months
Negative features	Additional releases of water
Costs	Minimal (lost hydropower revenue).
Applicability to reservoirs	Very applicable to projects with multiple release structures.

4.4.4 CONCENTRATION OF FLOW THROUGH ONE GATE

4.4.4.1 Description

Navigation dams are designed primarily to maintain a pool for navigation above the project and regulate flow for navigation downstream of the project. During the summer and early fall, typically the low-water period, flows through these structures may be minimal. These low flows may allow some thermal or chemical stratification to occur in the upstream pool, somewhat similar to a storage reservoir. Spill operations may not reaerate this poor-quality bottom water, which may lower downstream water quality.

The operation of a multi-bayed structure, such as a navigation dam, usually requires that all gates be operated the same for a given release condition to achieve a laterally-uniform flow distribution downstream. This type of operation is designed to prevent scour and erosion in the stilling basin and tailrace or to provide the best flow conditions for navigation. While hydraulic conditions may require uniform operation during high-flow events to minimize scour and erosion downstream, low-flow conditions may permit the concentrating of flow into a few gates.

During these low-flow periods, the dissolved oxygen may be low due to reservoir-type conditions in the upstream pool. Therefore, a modification of the gate operation during low-flow periods to increase the unit discharge may be enhance release dissolved oxygen (DO) by concentrating the flow to pass through a minimal number of gates (perhaps only one). This would increase turbulence and air entrainment in the stilling basin and enhance reaeration. As an example, a structure with six gates may operate with all 6 gates open at 0.5 ft, but this would like provide very little reaeration in the stilling basin. Modifying the operation to a single gate with a 3.0-ft opening concentrates the flow through and will likely increase turbulence and reaeration downstream (Figure 4.4.4).

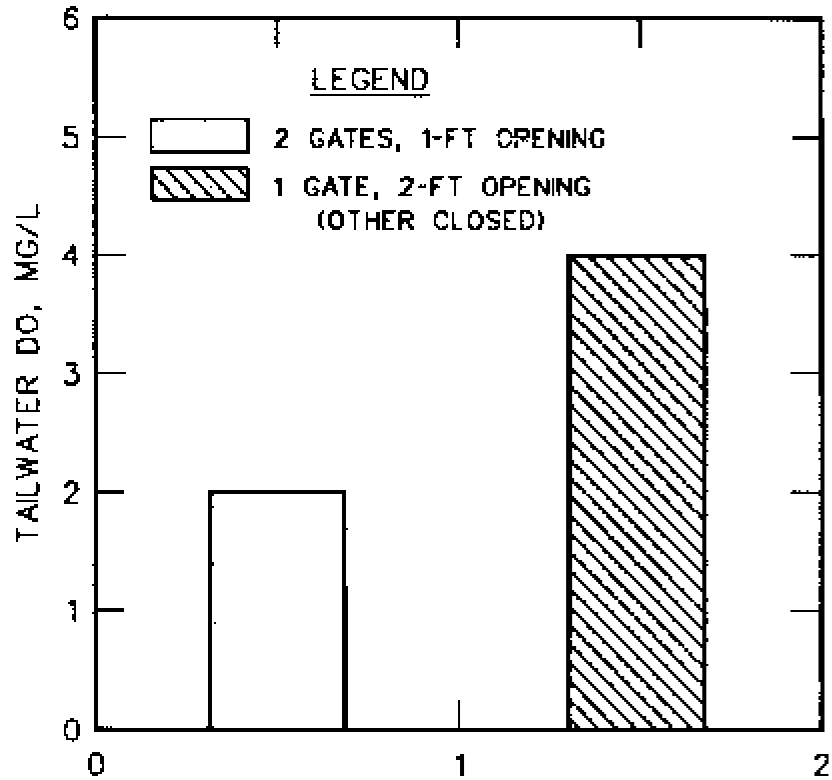


Figure 4.4.4. Example of effect of flow concentration through a single gate

4.4.4.2 Evaluation Methodology

Design Criteria. Concentrating flow through a small number of gates, as an operational alternative, can be implemented at nearly any structure where multiple gates extend horizontally across the dam. However, its application must be determined on a case-by-case basis. The specific objective of this technique is to increase gas transfer in the stilling basin of the spillway, usually to improve the dissolved oxygen, although other volatile solutes are also transferred. Although this technique has not been widely used, it has been applied at navigation dams on the Ohio River (Price 1990). A general method to evaluate the application of this technique is given below.A field investigation is the most effective way of defining the gas transfer characteristics of a hydraulic structure and the potential for concentrating flow. In a preliminary assessment, water temperature and oxygen concentration should be measured upstream and downstream of the project. If the upstream DO is near saturation, measurements of oxygen uptake across the structure will reveal very little about the gas transfer characteristics and nothing will be gained by implementing this technique. If, however, the upstream oxygen is low (more than 2.5 mg/l below saturation), then a field investigation will reveal the applicability of concentrating flow.

To determine the oxygen uptake characteristics, upstream and downstream DO concentrations should be measured for a range of spillway discharges. We recommend that recording instruments be deployed in an array that covers the immediate tailrace of the spillbay. These data will specifically show the effects of increasing discharge through a single spillbay. The specific operational characteristics of the structure should also be determined during the field investigation. This particularly includes the stilling basin performance to note changes in the submergence of the discharge jet or in air entrainment in the stilling basin. This experimental method was developed by Wilhelms (1988) at several lock and dams on the Ouachita and Red Rivers in Louisiana and has been used at many projects (Hettiarachchi, et al 1997; Portland and Walla Walla Districts, Corps of Engineers).

If the field investigation shows an improvement in downstream DO with increasing discharge per spillbay, then hydraulic concerns should be addressed prior to full implementation. Since most stilling basins and energy dissipators were designed for equal gate operations, unequal gate operation may create scour or erosion problems downstream. The operating limits for this technique must be determined.

The reaeration that is occurring with the existing gate operation can be compared to generalized relationships developed for some types of hydraulic structures given by Wilhelms (1988), ASCE (1990), and Gulliver, Wilhelms, and Parkhill (1998). Some research suggests that as the unit discharge increases, a decrease in reaeration may occur (Thene, Daniil, and Stefan 1989).

4.4.4.3 Implementation

Several of the lock and dams on the Ohio River are operated to maximize the unit discharge through a few gates and thereby increase DO downstream (Price 1990). Field investigations at Montgomery Lock and Dam on the Ohio River indicated that a 2-ft gate opening on one gate achieved 1.5 times the DO downstream as two gates open 1 ft (US Army Engineer District, Pittsburgh 1975).

4.4.4.4 Summary

Concentration of flow through a small number of spillbays is a technique that can improve DO in releases from low-head navigation dam spillways. While it could be used at almost any facility that operates a number of gates across the face of the structure, it is especially useful at gated-sill types of projects. The method relies on increasing the turbulence and air entrainment of the flow plunging into the stilling basin of the spillway.

Although reported applications of this technique are limited, it holds much potential for increased flow aeration with little or no structural modification. Information concerning this technique is summarized in Table 4.4.4.

Table 4.4.4

Summary of Concentration of Flow Through One Gate

Characteristics	Description
Targets	Low dissolved oxygen levels (and associated concerns)
Mode of action	Reaeration of flow through increased turbulence as a result of concentrating the flow through a minimum number of release gates
Effectiveness	Moderate
Negative features	Possible increased scour and erosion of stilling basin and downstream
	Change in the flow conditions and patterns for ship traffic
Costs	Minimal
Applicability to reservoirs	Most applicable to lock and dam projects but can be used at other projects where thp flow is usually spread out over multiple outlets

4.4.5 OPERATIONAL OPTIMIZATION FOR WATER QUALITY

4.4.5.1 Description

Reservoirs are operated for many purposes, most commonly flood control, hydropower, navigation, recreation, and water quality. At times, conflicting results may occur, when operating a structure for two or more of these purposes. For example, during the stratification season, the operational purpose is to provide cool water temperatures to the downstream, but degradation of the hypolimnetic oxygen content may result in low-dissolved oxygen (DO) water being released. In another example, hypolimnetic releases during the early portion of the summer exhaust the supply of cold hypolimnetic water, resulting in warm water releases that may be detrimental to the downstream fishery. Guidance can be developed with numerical optimization techniques to avoid or minimize these conflicts and deliver the desired water quality to the river and still meet other project purposes.

This technique relies on numerical water quality modeling to predict the release water quality under a variety of operational scenarios. With a multilevel withdrawal structure and a release water quality objective stated as a numerical function over time, a water quality model is used to predict in-reservoir and release quality over time. From these predictions, a measure of achieving the release objective for each operation (different release or withdrawal location) can be computed. Several operational scenarios are simulated iteratively and systematically using optimization techniques to compute how well each operational scenario meet the water quality objective. The operational scenario that most closely matches the water quality objective is the optimum operation for the period of concern. Figure 4.4.5 shows the release temperatures from a selective withdrawal tower based upon a “best daily” operation and then a “seasonal” operation. Allowing a small violation of the temperature objective in the early part of the year prevents a major violation in the late summer.

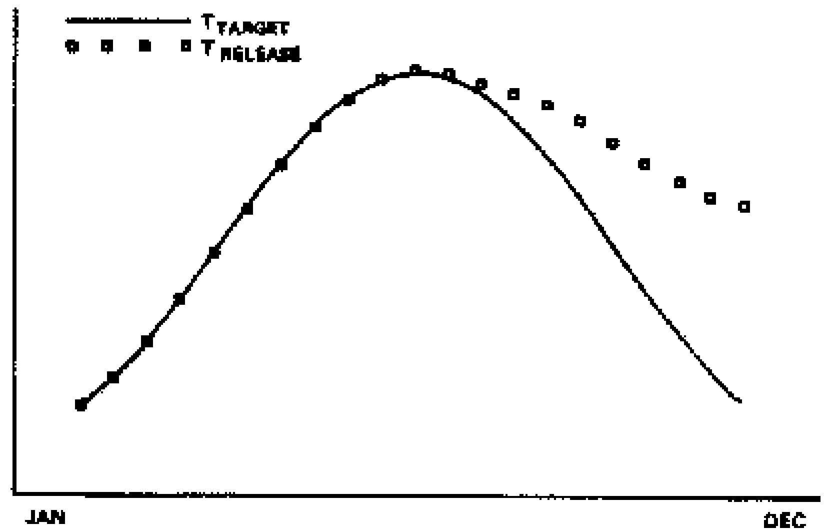
4.4.5.2 Design Criteria

Optimization techniques have been used in a variety of modeling applications, particularly in water supply investigations and design and operation of selective withdrawal structures. Although the techniques can become complicated because of the number of water quality objectives, the general approach remains the same as for a one-parameter optimization process.

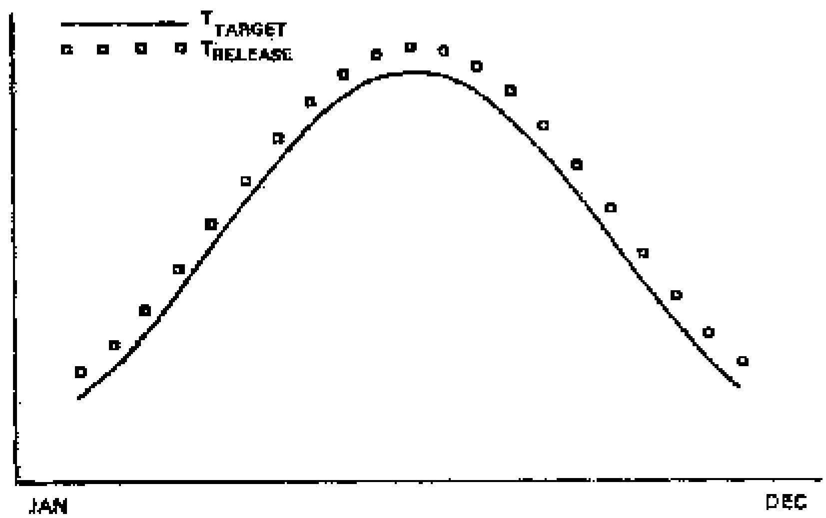
This approach for the optimization technique is outlined below.

a. Identify release water quality concern(s). This must be definable in scalar terms, such that an objective function can be formulated for use in a numerical model. For example, if the objective is to increase the release temperature during the fall period, the temperature objective must be defined in specific increments over time.

b. Identify release operation concerns. This will include the number of ports, their center-line elevations, heights, widths, and minimum and maximum discharges. If the reservoir is operated under a guide curve, this must also be considered prior to optimization.



a. Daily operation



b. Seasonal operation

Figure 4.4.5 Release temperatures from a selective withdrawal tower based upon a "best daily" operation and then a "seasonal" operation

c. Select a numerical procedure (coupling simulation and optimization) for simulation of the impacts of the operational scenarios on temporal water quality. Although the optimization process could be implemented by hand, using a numerical model for simulation of operational scenarios, it is most often implemented as a part of the numerical model, thereby allowing a computer routine to select the optimum scenario based on the water quality objectives specified. A detailed description of an optimization algorithm and its associated parameters is given in Poore and Loftis (1983), and application is given in Dortch and Holland (1984). The numerical model selected should accurately simulate the parameter(s) specified in the release water quality objective(s). The model must be adjusted and verified to ensure that predictions are accurate.

d. Conduct optimization simulations, using all viable operational scenarios to identify the optimum operation. Since meteorological conditions will play an important role in the reservoir water quality, extreme as well as average meteorological conditions should also be simulated.

This general procedure can be used to develop operational procedures as well as used in a real-time operational mode.

4.4.5.3 Implementation

The most extensive use of optimization techniques for water quality enhancement has been in the area of selective withdrawal structure design. Numerous investigations using this technique have proven effective for these purposes, particularly for an increase in reservoir storage for water supply (Holland 1982, Schneider and Price 1988, Price and Holland 1989).

Wilhelms and Schneider (1986) describe an optimization technique using a one-dimensional thermal model to operate a project to minimize temperature deviations over a long period of time. This investigation assumed that small deviations over a long period of time were more acceptable than large deviations over a short period of time.

Kaplan (1974) used a water quality index composed of several parameters linked to a reservoir water quality model to optimize reservoir releases for both downstream and in-lake water quality. His research concluded that the technique could be used to operate a reservoir to meet both flood control and water quality objectives with a selective withdrawal structure.

Others have extended the optimization technique to assist in operation of multi-reservoir systems. Fontane and Labadie (1982) developed a methodology for optimizing water storage strategies in the West. This approach used an optimization simulation model to locate water supply reservoirs while considering constraints such as water quality, flood control, and minimum flow needs.

Fontane, Labadie, and Loftis (1982) developed an objective-space dynamic programming tool to determine the optimum reservoir operational for long-term operation. This tool involved minimizing or maximizing the deviation between some in-reservoir or release water quality and a predetermined water quality criteria.

Howington (1989) used this technique to develop an operational strategy for the Lost Creek intake structure. In this investigation, the objective was to minimize the deviation between the release water temperature and downstream temperature target.

Bonazountas and Camboulives (1981) investigated the use of optimization in operating a series of reservoirs with flood control and water quality constraints. Their investigation used three objective functions and involved a complicated hierarchical optimization process to arrive at operating conditions.

Willey, Smith, and Duke (1985) developed a reservoir system analysis model to simulate water quality within a large reservoir system. This model, which can simulate up to 10 reservoirs with eight water quality parameters, uses a linear programming algorithm to determine operating conditions for a system of reservoirs with user-specified control points. It has been applied to the Sacramento River system, which has five reservoirs.

4.4.5.4 Summary

Operation of a multipurpose reservoir may lead to conflicting impacts on the water quality of the project. For example, operation of hydropower from a deep stratified reservoir may result in a discharge with low dissolved oxygen. If operational alternatives exist, such as selective withdrawal capabilities or the capability is under design, optimization techniques may provide operational or design guidance to minimize deviations from project objectives. This technique requires the coupling of a numerical model with mathematical optimization techniques to conduct simulations of reservoir operations on release.

Applications of this technique are found primarily in the operation of multilevel selective withdrawal outlet structures. Several examples of applications to reservoir projects were presented. Table 4.4.5 presents a summary of information on this technique.

Table 4.4.5

Summary of Release Strategy Optimization

Characteristic	Descriptions
Targets	Water quality of the reservoir as well as down-stream releases
Mode of action	Numerical modeling of reservoir operations to optimize the quantity and or quality of reservoir releases
Effectiveness	Dependent on reservoir storage capacity and objectives
Longevity	Seasonal
Negative features	Conflicting release requirements may reduce some operations in order to benefit others
Costs	Minimal

4.4.6 REFERENCES

Akiyama, J., and Stefan, H. 1985. "Turbidity Current with Erosion and Deposition," *Journal of Hydraulic Engineering*. Vol III, No. 12, pp 1473-1496.

ASCE 1990. "Gas Transfer at Hydraulic Structures," *Air - Water Mass Transfer: Selected Papers from the Second International Symposium on Gas Transfer at Water Surfaces*, Wilhelms and Gulliver, eds, American Society of Civil Engineers, pp 460-493.

Bonazountas, M., and Camboulives, J. M. 1981 (May). "Multi-Decision Analysis for Large-Scale River-Basin Reservoir Systems," *Journal of Hydrology*, Vol 51, No. 1-4, pp 139-149.

Davis, J. E., Holland, J. P., Schneider, M. L., and Wilhelms, S. C. 1987. "SELECT: A Numerical, One-Dimensional Model for Selective Withdrawal," Instruction Report E-87-2, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Dortch, M. S., and Holland, J. P. 1984. "A Technique to Optimally Locate Multilevel Intakes for Selective Withdrawal Structures," Technical Report HL84-9, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Findley, D. I., and Day, K. 1987 (Jul). "Dissolved Oxygen Studies Below Walter F. George Dam," *Proceedings: CE Workshop on Reservoir Releases*, Miscellaneous Paper E-87-3, US Army Engineer Waterways Experiment Station, Vicksburg, MS, pp 71-76.

Fontane, D. G., and Labadie, J. W. 1982. "Methodology for Developing Optimal Water Storage Strategies," International Symposium on Hydrometerology, June 13-17, American Water Resources Association, pp 257-261.

Fontane, D. G., Labadie, J. W., and Loftis, B. 1982 (Feb). "Optimal Control of Reservoir Discharge Quality Through Selective Withdrawal; Hydraulic Laboratory Investigation," Technical Report E-82-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Goranflo, H. M., Jr. , and Adams, J. S. , Jr. 1987 (Jul). "Providing Minimum Flows Below Hydropower Projects," *Proceedings: CE Workshop on Reservoir Releases*, Miscellaneous Paper E-87-3, US Army Engineer Waterways Experiment Station, Vicksburg, MS, pp 77-84.

Gulliver, J. S., Wilhelms, S. C., and Parkhill, K. L. 1998 (Jun). “Predictive Capabilities in Oxygen Transfer at Hydraulic Structures,” *Journal of Hydraulic Engineering*, American Society of Civil Engineers, Vol 124, No. 6.

Hettiarachchi, S. L., et al 1997. “Reaeration Measurements Using a Methane Tracer at Six Dams in the Ohio River Basin,”

Headquarters, US Army Corps of Engineers. 1987a. "Management of Water Control Systems," Engineer Manual 1110-2-3600, US Government Printing Office, Washington, DC.

Headquarters, US Army Corps of Engineers. 1987b. "Reservoir Water Quality Analyses," Engineer Manual 11102-1201, US Government Printing Office, Washington, DC.

Hogan, K. B. 1986. "Development of Reservoir Operating Rules with Water Quality Objectives," Ph.D. dissertation, Johns Hopkins University, Baltimore, MD.

Holland, J. P. 1982. "Effects of Storage Reallocation on Thermal Characteristics of Cowanesque Lake, Pennsylvania; Numerical Model Investigation," Technical Report HL-82-9, US Army Engineer Waterways Experiment Station, Vicksburg, Ms.

Howington, S. E. 1989. "Intake Structure Operation Study, Lost Creek Dam, Oregon," Technical Report HL-89-13, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Iwasa, Y. and Matsuo, N. 1981. "Estimation of Turbidity in Reservoirs," Proceedings of International Association for Hydraulic Research. New Delhi, India.

Kaplan, E. 1974. "Reservoir Optimization for Water Quality Control," Ph.D. thesis, Pennsylvania University, Philadelphia, PA.

Murota, A. and Michioku, K. 1986. "Analysis on Selective Withdrawal from Three-Layered Stratified Systems and Its Practical Application," Journal of Hydroscience and Hydraulic Engineering. Vol. 4(l). pp. 31-50.

Newman, B. A., and Loucks, D. P. 1975. "Derivation and Evaluation of Reservoir Rule Curves," Report No. W76-04014, Office of Water Research and Technology, Washington, DC.

Nix, J. 1981. "Contribution of Hypolimnetic Water on Metalimnetic Dissolved Oxygen Minima in a Reservoir," Water Resources Research. Vol. 17, No. 2. pp. 329-332.

Poore, A. B., and Loftis, B. 1983. "Water Quality Optimization Through Selective Withdrawal," Technical Report E-83-9, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Price, R. E. 1990. "Water Quality Enhancement Techniques Used Within the Corps of Engineers," Technical Report E-90-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Price, R. E., and Holland, J. P. 1989. "Thermal Analysis of Prompton Reservoir Modification, Pennsylvania," Technical Report HL-89-20, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Schiebe, F. R., and Dendy, F. E. 1978. "Control of Water Residence Time in Small Reservoirs," *Transactions of the American Society of Agricultural Engineers*, Vol 21, No. 4, pp 666-670.

Schneider, M. L., and Price, R. E. 1988. 'Temperature Analysis, Howard A. Hanson Reservoir, Washington; Mathematical Model Investigation," Technical Report HL-88-22, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Thene, J. R., Daniil, E., and Stefan, H. G. 1989 (Feb). "Effect of the Hastings Hydropower Project at Lock and Dam No. 2 on Dissolved Oxygen in the Mississippi River," Project Report No. 281, University of Minnesota, St. Anthony Falls Hydraulic Laboratory, Minneapolis, MN.

US Army Engineer District, Little Rock. 1985 (Feb). "Dissolved Oxygen Study, Table Rock Dam and Lake, White River, Missouri," Little Rock, AR.

US Army Engineer District, Pittsburgh. 1975 (Jun). "An Evaluation of the Effects of Mainstream Navigation Dams on the Water Quality of the Upper Ohio River," US Army Engineer District, Pittsburgh, Pittsburgh, PA.

US Army Engineer Districts, Portland and Walla Walla. 1996. “Dissolved Gas Abatement: Phase I,” Technical Report, US Army Engineer District - Portland, Portland, OR.

Van Pagee, J. A., Groot, S., Klomp, R., and Verhagen, J. H. G. 1982. "Some Ecological Consequences of a Projected Deep Reservoir in the Kabalebo River in Suriname," *Hydrobiological Bulletin,* Vol 16, No. 2-3, pp 241-254.

Wilhelms, S. W. 1988 (Jul). "Reaeration at Low-Head Gated Structures: Preliminary Results," Information Exchange Bulletin E-88-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Wilhelms, S. C., and Schneider, M. L. 1986. "Operational Tools: Optimal Control of Reservoir Water Quality," *Proceedings: CE Workshop on Design and Operation of Selective Withdrawal Intake Structures*, Miscellaneous Paper HL86-3, US Army Engineer Waterways Experiment Station, Vicksburg, MS, pp 110 - 115.

Wilhelms, S. C., and Smith, D. R. 1981. 'Reaeration Through Gated-Conduit Outlet Works," Technical Report E-81-5, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Willey, R. G., Smith, D. J., and Duke, J. H. 1985 (Feb). "Modeling Water Resources Systems for Water Quality," Report No. HEC-TP-104, Hydrologic Engineering Center, Davis, CA.

4.5 IN-RESERVOIR TECHNIQUES: AERATORS/OXYGENATION

4.5.1 DESCRIPTION

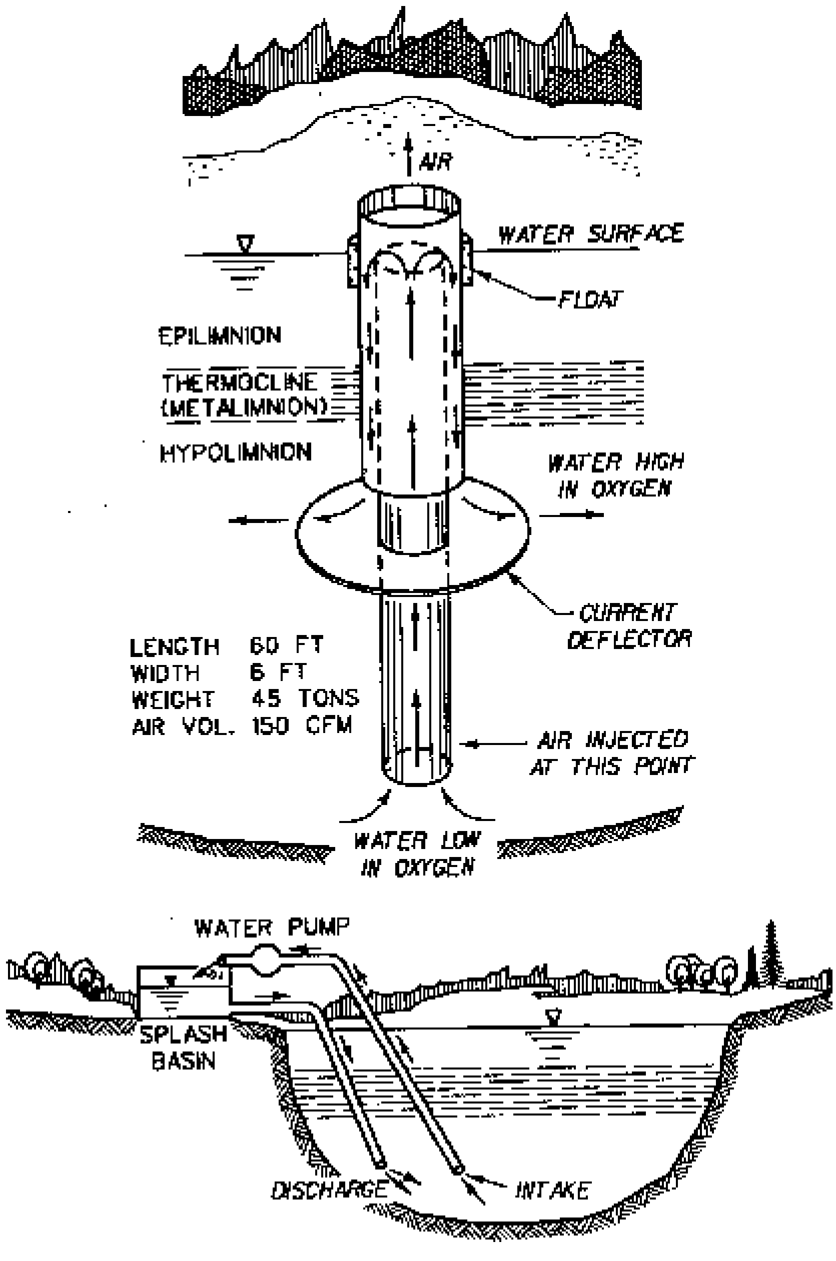
The thermal stratification that occurs in most reservoirs isolates the hypolimnion from circulation and oxygen absorption at the surface. Because of hypolimnetic oxygen degradation, the hypolimnion may become anoxic or anaerobic, and allow oxidized elements and compounds to be reduced into solution. These constituents, such as iron, manganese, and hydrogen sulfide, are toxic to aquatic life and, along with the low or zero oxygen content, may pose an in-reservoir or downstream water quality problem.

An aeration system can improve the low dissolved oxygen (DO) content of a reservoir or lake hypolimnion by directly introducing air or molecular oxygen into the water column. These systems usually consist of an oxygen or air source, supply lines, and a diffuser system to generate bubbles. An aeration system can be an open, free bubble column in the reservoir or confined within a floating or submerged system. The selection of the aeration system type depends upon the objectives and constraints under which the system will operate.

An aeration system may destratify the reservoir by mixing the hypolimnion and epilimnion. This technique may eliminate the anoxic conditions, but will also warm the hypolimnetic water to a temperature near that of the epilimnion. If cool water is needed for a cold-water fishery, then the thermal stratification has to be maintained. Thus, the hypolimnion must be aerated without impacting the thermal stratification.

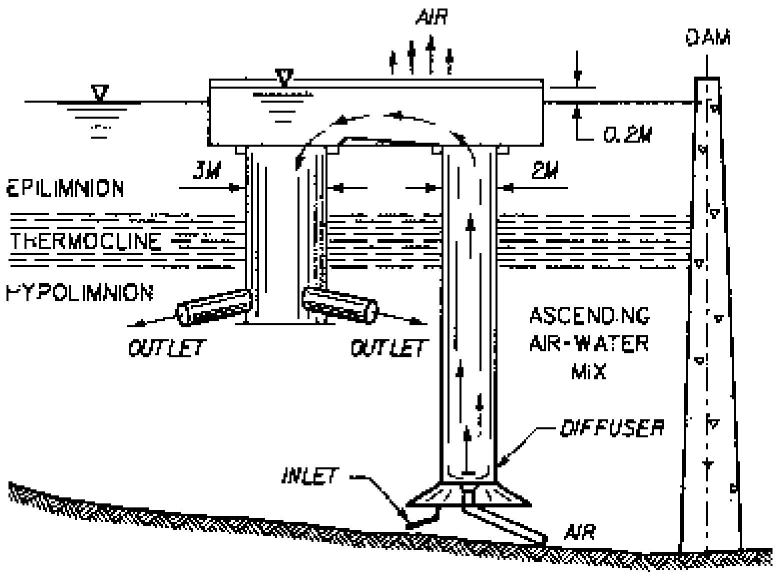
4.5.2 THEORY

Hypolimnetic aeration/oxygenation can be accomplished through a variety of approaches, ranging from pumping the hypolimnetic water to the surface for aeration and returning it to the hypolimnion, to fine-pore pneumatic diffusers placed in the hypolimnion for the introduction of gaseous oxygen (Figure 4.5.1). These systems use either air or liquid oxygen as the oxygen source. The volume of oxygen required for reaeration of the hypolimnion is based on the volume of the hypolimnion and the hypolimnetic oxygen degradation rate. Pneumatic diffuser systems are designed such that the rising bubble plume does not mix hypolimnetic water into the epilimnion and thus begin destratification. Systems that pump hypolimnetic water to the surface for reaeration are designed to minimize the velocity of the return flow so that mixing of the return flow does not induce destratification.

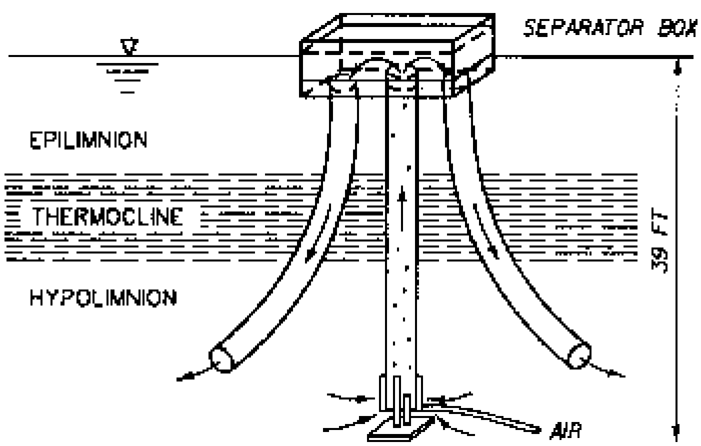


a. Full airlift hypolimnetic aeration used in Hemlock Lake, Michigan

Figure 4.5.1 Example hypolimnetic aeration devices



b. Full airlift hypolimnetic aerator, used in Wahnbach Reservoir, West Germany



c. Full airlift hypolimnetic aerator, used in Mirror and Larson Lakes, Wisconsin

Figure 4.5.1 Example hypolimnetic aeration devices

4.5.3 DESIGN METHODOLOGY

The design of an aeration or oxygenation system should begin with a clear definition of the reservoir thermal and dissolved oxygen stratification characteristics. If the hypolimnetic DO concentration drops below an acceptable level relative to the project objectives, then the aeration/oxygenation system would be activated to maintain in-reservoir or release oxygen concentrations above prescribed minimums. The general methodology described below is adapted from Lorenzen and Fast (1977). Modifications have been made to include developments since 1977.

a. Determine the hypolimnetic volume. Using historical thermal profiles during the summer months, the highest elevation of the hypolimnion must be determined. Then, using a depth-volume curve, the volume of water residing in the hypolimnion can be determined.

b. Estimate the hypolimnetic oxygen degradation (HOD) rate. Once thermal stratification has begun, the hypolimnetic DO content is measured periodically until the bottom DO concentration reaches 1.0 mg/L. The DO for each sample period is plotted against time with the slope of the resulting curve representing the HOD. The empirical model PROFILE (Walker 1987) can be used to compute the HOD and provide an analysis of various thermocline elevations on the HOD. The HOD rate is the minimum rate at which oxygen can be supplied to maintain aerobic conditions in the hypolimnion. The determination of the rate of oxygenation that the aeration system should be designed for should be made based on the desired DO concentration to be maintained in the hypolimnion.

c. Determine the type of system to be designed. Since the depth of the aeration system, as well as the gas flow rate, will impact the system efficiency, specific design criteria will be dependent on the type of system selected. One system may inject oxygen for release from the reservoir, while a different system might diffuse oxygen for in-reservoir improvement. Some manufacturers of aeration devices supply the air/oxygen delivery rate for their products and should be contacted for gas delivery rates and oxygen absorption efficiency. But the latest design technology uses materials not usually associated with air/oxygen injection.

4.5.4 IMPLEMENTATION

Fast and Lorenzen (1976) compiled an overview of hypolimnetic aeration system designs and experiences. They describe the three basic designs: mechanical agitation, oxygen injection, and air injection. Fast, Lorenzen, and Glenn (1976) also provided comparative costs for various hypolimnetic aeration devices and concluded that each approach must be evaluated relative to the aeration site to determine construction and operation costs. Pastorok, Lorenzen, and Ginn (1982) provide a review of hypolimnetic aeration and oxygenation experiences from 15 case studies along with theoretical aspects and impacts on various components in a lake. The Tennessee Valley Authority conducted tests of an oxygen injection system at Fort Patrick Henry Dam to determine an optimum configuration to improve hydropower releases (Fain 1978). Various types of diffusers and configurations were tested, achieving

98-percent peak efficiency in oxygen absorption. Toetz, Wilhm, and Summerfelt (1972) provided a review of the impacts of aeration/oxygenation systems on biological components along with an extensive bibliography of selected abstracts.

The hypolimnetic aeration or oxygenation cases referenced above were on a variety of reservoirs. Since then, oxygenation systems have been designed and installed at several reservoirs in the Tennessee Valley Authority and Corps of Engineers. These systems have been designed to increase release DO and minimize mixing between the hypolimnion and epilimnion. An oxygenation system for the Richard B. Russell Reservoir on the Savannah River in Georgia was designed to deliver 150 tons of oxygen per day to maintain release DO at 6 mg/L in the hydropower releases without increasing the release temperature (USAED, Savannah 1981; Gallagher 1984). Even more recently, TVA developed an innovative system using ordinary garden soaker hoses as the delivery diffuser and flexible ABS piping for a supply and floatation system (Mobley and Brock 1996).

4.5.5 SUMMARY

Aeration or oxygenation is a technique designed to increase the in-reservoir and release oxygen concentration. Some designs can accomplish this without impacting the thermal stratification. Other designs because of vertical mixing induced by the aeration system may partially destratify the reservoir. In most cases, the objective is to add oxygen without creating mixing cells in the lake that induce destratification. The design approach consists of defining the hypolimnetic oxygen degradation rate, the volume of water requiring treatment, and the amount of oxygen required to meet in-reservoir or release oxygen objectives. The actual system selection will depend upon the operational objects. Several aeration/oxygenation systems have been installed in Bureau of Reclamation, TVA, and CE projects. The most recent installations have been at TVA reservoir using a relatively inexpensive porous-hose delivery system. Summary information on aeration/oxygenation systems is presented in Table 4.5.1.

Table 4.5.1

Summary of Aeration-Oxygenation Systems

Characteristic	Description
Targets	Anoxic or low dissolved oxygen conditions (and associated concerns) in the reservoir as well as the release water without destratifying the reservoir.
Mode of action	Injection of oxygen or air into the lower reaches of the reservoir to increase the dissolved oxygen levels.
Effectiveness	Very.
Longevity	Years.
Negative features	Air systems require approximately five times the volume gas compared to pure oxygen systems
	High maintenance levels required for diffuser lines.
	Partial destratification possible, especially with higher volume gas flow rates if the diffusers are not spaced properly.
Costs	High (installation and maintenance).
Applicability to reservoirs	Very applicable to reservoirs with anoxic or low-DO conditions.

4.5.6 REFERENCES

Fain, T. G. 1978 (Oct). "Evaluation of small-pore diffuser technique for reoxygenation of turbine releases at Fort Patrick Henry Dam," Report No. WM28-1-32-100, Tennessee Valley Authority, Norris, TN.

Fast, A. W., and Lorenzen, M. W. 1976 (Dec). "Synoptic survey of hypolimnetic aeration," *Journal of Environmental Engineering*, ASCE, Vol 102, No. EE6, pp 1161-1173.

Fast, A. W., Lorenzen, M. W., and Glenn, J. H. 1976 (Dec). "Comparative study with costs of hypolimnetic aeration, *Journal of Environmental Engineering*, ASCE, Vol 102, No. EE6, pp 1175-1187.

Gallagher, J. W., Jr. 1984 (Jan). "Richard B. Russell Dam and Lake oxygen injection system," *Proceedings of a Seminar on Applications in Water Quality Control*, US Army Corps of Engineers, Committee on Water Quality, Washington DC.

Lorenzen, M., and Fast, A. 1977 (Jan). "A guide to aeration/circulation techniques for lake management," EPA-600/3-77-004, Ecological Research Series, US Environmental Protection Agency.

Mobley, M. H. and Brock, W. G. 1996. "Aeration of reservoirs and releases using TVA porous hose line diffuser," *Proceedings: ASCE North American Congress on Water and Environment*, Anaheim, June 22-28, 1996.

Pastorok, R. A., Lorenzen, M. W., and Ginn, T. C. 1982. "Environmental aspects of artificial aeration and oxygenation of reservoirs: a review of theory, techniques, and experiences," Technical Report E-82-3, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Toetz, D., Wilhm, J., and Summerfelt, R. 1972. "Biological effects of artificial destratification and aeration in lakes and reservoirs–Analysis and bibliography," Report No. REC-ERC-72-33, Oklahoma Cooperative Fishery Unit.

US Army Engineer District, Savannah. 1981 (Feb). "Design Memorandum No. 3, Richard B. Russell Dam and Lake, General Design Memorandum, Supplement No. 2, Water Quality Control System," Savannah, Ga.

Walker, W. W., Jr. 1987. "Empirical methods for predicting eutrophication in impoundments; Report 4, Phase III: Applications Manual," Technical Report E-81-9, US Army Engineer Waterways Experiment Station, Vicksburg., MS.

4.6 IN-RESERVOIR TECHNIQUES: MIXERS

4.6.1 DESTRATIFICATION

4.6.1.1 Problem Addressed

The thermal stratification that develops in most reservoirs during the summer months can isolate the hypolimnion from the epilimnion and block the transfer of oxygen from the surface to the bottom waters. This can result in anoxic conditions in the lower levels of the reservoir, which can expand as biological and chemical activity continues to deplete the dissolved oxygen (DO). Reservoir destratification is an enhancement technique in which hypolimnetic and epilimnetic water are mixed to eliminate or prevent stratification. If total reservoir destratification is not practical or desired, a system can be designed to affect a specific region of the lake, such as in front of intake structures to improve the water quality of the releases. Although destratification can be achieved with mechanical pumps, pneumatic systems have been the most commonly used systems in large reservoirs.

Destratification will maintain near-uniform mixed conditions through the water column. These conditions will impact in-reservoir and release water quality. For example, algal blooms can be reduced by limiting the amount of sunlight the algae receive (by circulating the algae to below the photic zone). Initial start-up of the system can increase nutrient loading and DO depletion if sediment is resuspended and exposed to upwelling water currents. Destratification of the reservoir will tend to reduce the habitat available for cold water fisheries, but increase the habitat for warm water fisheries. It will warm the release waters, which may impact the users downstream. Lorenzen and Fast (1977) outline some of the additional benefits and consequences for reservoir destratification.

4.6.1.2 Theory

Destratification of a specific portion of the lake or reservoir can be accomplished by the introduction of a diffused bubble plume in the water column. The rising bubbles induce a flow pattern by entraining hypolimnetic water transporting it to the surface, where it moves out laterally. Additional water moves in to replace the upward flow, and a circulation cell is created (Figure 4.6.1). The size and volume of these cells are usually determined by the physical boundaries of the reservoir, the strength of the generation source, and the configuration of the diffusers.

4.6.1.3 Design Procedures

Johnson (1984) provides a good overview of the various types of aeration/destratification systems in use and how they operate. Also provided are some guidelines for system selection based upon the lake characteristics and the goals of the system to be installed.

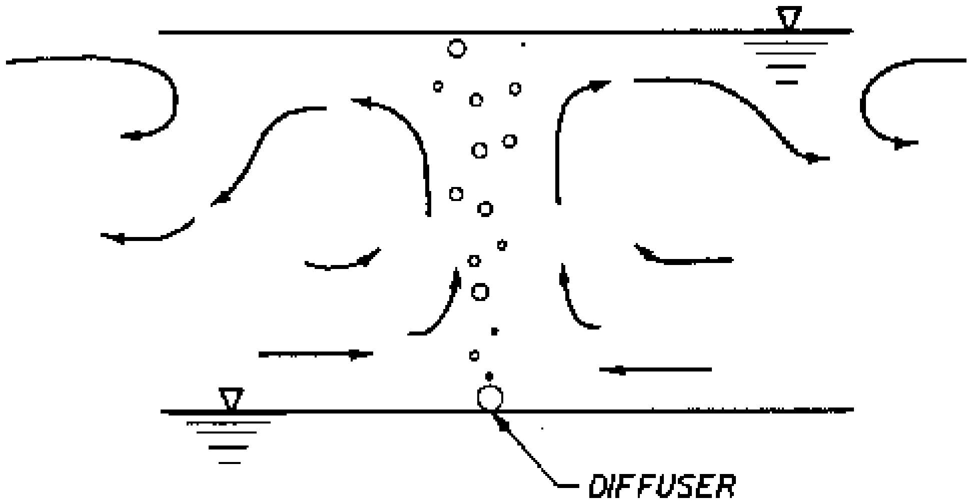


Figure 4.6.1 Destratification induced by bubble column

In general, the system design is based on the volume of water to be fully mixed, a representative temperature profile, and the stability of the reservoir (the potential energy of the mixed system minus the potential energy of the stratified system). The pneumatic-system must input enough energy into the reservoir to overcome the computed stability and thereby destratify the region of interest.

Davis (1980) also presented guidelines for a total or partial-lake destratification system. This design relies on a single perforated pipe (linear diffuser) to create a bubble curtain in the water column to induce vertical mixing. The guidance presented in the following paragraphs is a summary of the procedures outlined by Davis (1980) used for sizing a system to totally destratify a lake.

The first step in the design process is the computation of stability of the reservoir under a particular stratification. For a given temperature profile and reservoir bathymetry, the stability of the volume to be destratified, defined as the potential energy of the mixed system minus the potential energy of the stratified system, is determined. Davis recommends use of a temperature profile with a 4°C change through the thermocline for this process.

The total theoretical energy required to destratify the system is based on this stability and the solar and wind energy input to the reservoir volume. Davis (1980) provides guidelines for estimating the extent of solar and wind input into the system.

Next, the quantity (flow rate) of air required to destratify the reservoir volume is calculated based on the total theoretical energy, the time required to destratify the volume, and the depth of water above the perforated pipe. Davis (1980) recommends a minimum air flow rate of 20 L/sec.

A diffuser system that can accommodate the required air flow rate is then sized based on the volume, depth, air flow, and time to destratify. The diffuser system consists of a perforated (1-mm-diam holes located 0.3 m on center) linear system of pipe anchored to the bottom of the lake. The recommended minimum limit of perforated pipe length is 50 m (Davis 1980).

The compressor must be sized to provide the required air flow at a pressure sufficient to overcome the back-pressure and losses in the delivery and diffuser system, which include the hydrostatic pressure head, pressure loss due to friction and at the bends, and excess pressure at the end of the pipe. Davis (1980) provides the necessary equations to calculate all of the required pressures.

The free air flow through one diffuser hole is determined using a relationship between the ratio of absolute hydraulic pressure (press due to the water depth and atmospheric pressure) to the mean internal pressure (pressure at the end of the diffuser pipe plus the absolute hydraulic pressure). This value is multiplied by the number of holes in the diffuser pipe. The result should be equal to or greater than the air flow required. If not, the length of pipe must is increased and the pressure requirements a recalculated. Several iterations through the pressure and length calculation and checks may be required to arrive at a satisfactory solution.

To complete the system, an anchoring system that is capable of keeping the diffuser pipe submerged must be designed.

According to the design criteria (Davis 1980), the system should be activated at the beginning of the stratified period or when the oxygen content of the hypolimnion falls to 50 percent of the saturation level. Operation should continue throughout the stratification season to maintain isothermal conditions or the desired DO level.

4.6.1.4 Applications of Pneumatic Destratification

A number of pneumatic destratification systems have been installed in the United States England, Australia, and Europe. Summarized in Table 4.6.1 are reservoir characteristics and the pneumatic destratification system parameters for several systems that have been installed and tested.

Early equipment design and testing were conducted by the US Army Engineer District, Savannah, on the destratification system installed and operated in Allatoona Lake, Georgia (USAED, Savannah 1973). This configuration used a multiple-port diffuser system consisting of five cross-type diffusers (approximately 19.8 m across) clustered about 2,000 ft upstream of the dam face. Each diffuser was supplied with a 250-cfm compressor. This system was operated continuously from May through September 1968 and from mid-March through mid-September

Table 4.6.1											
Pneumatic Destratification Installations											
				Depth		Temperature[2]			Diffuser		
Source[1]	Reservoir Location	Volume m^3	Surface Area m^2	Avg. m	Max. m	T_e ºC	T_h ºC	Air Flow Rate L/sec	Type	Size	Remarks
1	"Hundred en dergtig"	32E6	2.2E6	13	27	19	13.5	200	Point		
1	"Pertrus plaat" '77	13E6	1.0E6	13	15	17	14	170	Point		
	'78	13E6	1.0E6	13	15	18	13	170	Point		
2	Tarago Res (Aus) '76	37.6E6	3.6E6	10.5	23	22	13	150 50	Linear	40 m	To start To maintain
	'77	37.6E6	3.6E6	10.5	23	14	12	50 100	Linear	40 m	To start To maintain
3	Cotter Res (Aus)	4.7E6			30.5	24	11	120	Point	6.1 m	Cross Type[3]
4	Kangaroo Res (S. Aus)	24.4E6	1.2E6	20.2	50	21	12.5	250	Linear	2-12.2 m	
4	Little Para (S. Aus)	21.4E6	1.5E6	14.2	49.5	17	12	300	Linear	2-15 m	
5	Prospect Res (Aus)	50.1E6	12.6E6	9.8	24	19.5	16	75 150	Linear	98 m	To start To maintain
6	Allatoona L.	453E6	48.0E6	9.4	45.7	26	15	590	Point	19.8 m	Cross type[3]

[1] Sources: 1 - Kranenburg 1979, 2 -Burns 1981, 3 - Smith 1981, 4 - Croome 1981, 5 - Bowen 1981, 6 - USAED, Savannah 1973.
[2] T_e and T_h refer to the average temperature of the epilimnion and hypolimnion, respectively.
[3] Cross type diffusers are measured from end-to-end of one arm.

1969. While the system didn't successfully destratify the entire lake, the system reduced the stratification in the area around the diffusers and in the vicinity of the dam. The extent of the zone of influence was difficult to determine; however, some data indicated that the system affected the DO levels as far as 4 miles upstream from the dam.

Other prototype experiments were conducted in 1976, 1977, and 1978 (Kranenburg 1979) using a point diffuser system. All of these tests were begun under stratified conditions and were operated and monitored for approximately 1 week. During the 1976 tests, the air flow rate was not sufficient to cause total lake destratification, but did succeed in affecting the portion of the reservoir in the area of the diffuser. There was minimal change in the temperature of the epilimnion, but the metalimnion and hypolimnion increased approximately 3ºC over time (assumed to be a result of vertical mixing).

In the 1977 and 1978 tests, the influence of the bubble plume was found to extend over the entire reservoir. The water temperature profile became fairly uniform top to bottom over the course of the tests, indicating substantial vertical mixing due to either the bubble plume and/or wind mixing. These tests were used to verify laboratory predictions of the destratification process and did not consider DO in the process.

Burns (1981) details some of the design and installation procedures used at Tarago Reservoir in Victoria, Australia. The design uses two 20-m lengths of pipe containing diffusers. Two operational configurations were tested (one in 1976 and the other in 1977), and both were successful in destratifying the majority of the reservoir. The 1976 test began with operation of the air system under strongly stratified conditions. A high air flow rate (150 L/sec) was used to destratify the reservoir (in 14 hr) initially. This was reduced to 50 L/sec to maintain the destratification. During the 1977 test, the system was started at the beginning of the stratification season. Initially, 50 L/sec was used, but as the season progressed the air rate had to be increased to 100 L/sec to maintain destratification.

The destratification system designed for the Cotter Reservoir as part of the water supply system for Canberra, Australia (Smith 1981) used a cross-shaped diffuser system, similar in design to the diffusers used at Allatoona Lake, Georgia, but smaller. This system was to be used to alleviate high iron and manganese problems experienced as the hypolimnion became anoxic through the summer. At the time of publication of the report, the system had been installed, and preliminary testing was taking place to determine optimum air flow rate and operating schedule. However, further information about the success of this system has not been published.

Croome (1981) describes two destratification systems used on Kangaroo Creek Reservoir and Little Para Reservoir, both in South Australia. Kangaroo Creek Reservoir was destratified twice, once in 1977 and again in 1978 under different stratification intensities. The system utilized two 12.2-m-long (100-mm-diam) galvanized steel pipes located approximately at the heel of the dam. Sixteen fine bubble dome diffusers were equally spaced along each length of pipe (32 diffusers total). During both tests, the reservoir was allowed to stratify before the system was started.

Destratification of the entire reservoir occurred in 15 days and significantly reduced the iron and manganese levels. The reservoir was allowed to begin to restratify afterwards. The effects on turbidity, algae, iron, manganese, dissolved oxygen, and nutrients were monitored.

A similar system was tested and monitored on the Little Para Reservoir, South Australia (Croome 1981). Two 15-m-long pipes, each with 20 diffusers (40 total), were located on the face of the dam near the level of the lowest intake port. Total destratification of the reservoir was accomplished in approximately 10 days. A smaller compressor (175 L/sec) will be installed permanently for continuous operation through the stratification season, beginning once the thermocline has established itself (or the hypolimnion begins to become anoxic).

Bowen (1981) examined a system installed on Prospect Reservoir, Sydney, Australia, to correct hypolimnetic oxygen deficiency. Rather than formed diffusers, holes were drilled in the pipe to act as diffusers (this system is similar to the design procedure proposed by Davis 1980). The system was started after stratification had been established and was stopped after the reservoir had become isothermal (about 30 days).

4.6.1.5 Summary

Pneumatic destratification can improve the water quality within and released from a reservoir. This technique uses a rising bubble plume to create vertical mixing. Various forms of pneumatic destratification systems have been installed with varying degrees of success. In general, linear diffusers strategically positioned on the lake bottom are supplied by a compressor located on the shore. The air bubbles released through the diffusers rise through the water column and create an up-welling water current that spreads out laterally upon reaching the surface. Circulation cells are set up to replace the upward moving water, and eventually the water column becomes mixed to the level of the diffuser(s). A method for designing a system was presented, along with references for computational procedures. Table 4.6.2 summarizes information concerning this technique.

4.6.2 LOCALIZED MIXING

4.6.2.1 Problem Addressed

The thermal stratification that occurs at most reservoir projects can create problems with release water quality. If the hypolimnion becomes isolated from the surface and turns anoxic, reducing conditions for iron, manganese, and hydrogen sulfide will prevail. If a hydraulic structure withdraws water from the hypolimnion, release of these soluble constituents as well as

Table 4.6.2

Summary of Pneumatic Destratification

Characteristic	Description
Targets	Reservoir conditions that are initiated or aggravated by thermal stratification or that may be improved by mixing (i.e. anoxic conditions, nutrient cycling, chemical resuspension, algal blooms, etc.).
Mode of action	A continuous bubble stream is used to create recirculation cells in the reservoir to mix or circulate hypolimnetic and epilimnetic waters.
Effectiveness	Very effective.
Longevity	Months to years.

low DO could create water quality problems downstream. Localized mixing is a technique that is being used as a means to improve the release quality of some projects which exhibit these characteristics. It can also be used to improve the water quality in localized areas in the reservoir by mixing epilimnetic with hypolimnetic waters.

4.6.2.2 Theory

Localized mixing is a technique that provides enough mixing in a local area to reduce, if not eliminate, the thermal stratification. This is usually accomplished by pumping surface water down into the hypolimnion to achieve a locally uniform vertical temperature profile (Figure 4.6.2). The jet from the pump provides the energy needed to disrupt the stratification and causes entrainment of the epilimnetic waters in the release discharge to enhance the water quality.

This technique, as indicated by its name, is used to mix only a small or localized area of the reservoir. Therefore, only the area of concern as related to the localized mixing objective is impacted. In the case of improvement of release quality, the intended impacts are related only to the release volume, not the entire reservoir. If this technique were used to improve the water quality of a small cove or marina, only the area under consideration (such as the cove or marina) enters into the design methodology. This also minimizes costs of the system.

4.6.2.3 Design Methodology

The design of a localized mixing system will usually involve determination of the desired water quality to be achieved in the release. A general procedure for design of a system was given by Holland (1983). The procedure described below is modified from Holland (1983) to include recent developments.

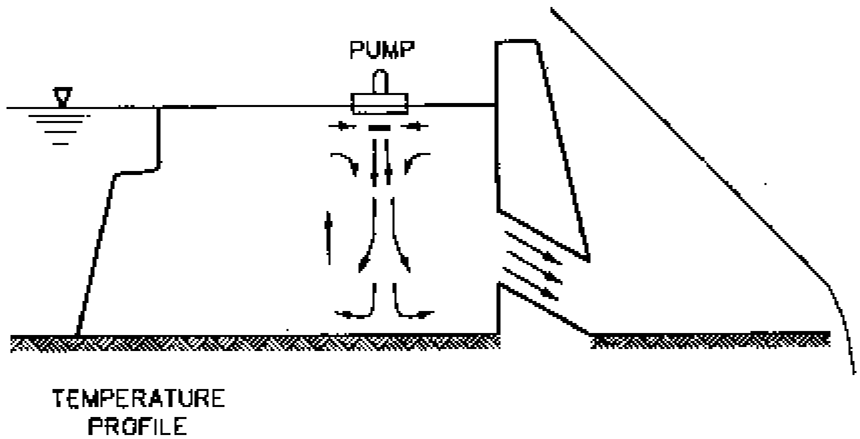


Figure 4.6.2 Schematic of localized mixing application

a. The specific objective of the localized mixing application must be determined, such as improvement of release water quality. This will involve determination of release temperature, dissolved oxygen, and/or trace constituents such as iron, manganese, and hydrogen sulfide in the existing release water. The desired release quality and the vertical profile of the water quality in front of the intake must then be determined. This involves the parameter(s) defined above as the specific objective. If the desired release quality does not exist in the profile taken in the reservoir, localized mixing will not meet the objectives.

b. Assuming that the desired release quality exists in the reservoir, the next task is the determination of the withdrawal zone characteristics of the intake structure. This determination is formulated through the use of the SELECT model (Davis et al. 1987). This model requires input of the thermal and quality profile of concern, release quantity, and structure intake configuration for computation of the vertical withdrawal profile and predicted release quality. Verification of the prediction is made through comparison with observed releases. The results of this model will indicate the relative proportions of water in the release coming from a given layer, as well as the limits of the withdrawal zone.

c. With the withdrawal description, the volume of the desired quality water needed to achieve the release quality goal should be determined. This can be determined using a simple mass balance criterion by diluting the release with epilimnetic water. The dilution factor, which is the ratio of epilimnetic volume to release volume, can be used in this determination. However, there are some limitations to the amount of dilution that can be achieved. According to Moon, McLaughlin, and Moretti (1979), the maximum dilution that can be achieved is approximately 75 percent. This means that for a total release volume of 1,000 cfs, only 750 cfs could be replaced by epilimnetic water.

d. The next task is the determination of the depth of penetration of the jet required to ensure release of the pumped water. This depth will usually be the center-line elevation of the intake. Computational methods for depth of penetration of a jet are given in Holland (1984).

e. The initial jet characteristics are to be determined next. Using computations of Albertson et al. (1950) and Holland (1984), the initial jet diameter and velocity can be determined. The selection of a pump to meet these criteria, which can be obtained from pump manufacturers, can then be made.

4.6.2.4 Applications of Localized Mixing

A number of applications of localized mixing have been made. Mechanical pumps used in these applications range from large-diameter axial flow pumps to relatively small-diameter direct-drive mixers. Limitations to the site specific application of surface water pumps include, temperature increase in tailwater, availability of sufficient electric power at the intake, and problems with trash and debris in the reservoir.

The earliest reported use of a pump to locally mix a reservoir for release improvement was Garton and Rice (1976). This work was based on a large-diameter (16.5-ft) axial flow pump used in lake destratification experiments on Lake Arbuckle in Oklahoma. The 16.5-ft-diam axial flow pump generated up to 207,000 gal/min of discharge And was successful in improving the release DO by 1 to 2 mg/L.

In 1977, localized mixing tests were conducted at Lake Okatibbee in Mississippi. These tests used a 1.83-m-diam axial flow pump designed to pump 1.7 m3/sec while the release structure released 1.4 M3/sec. Results of these tests indicated an improvement in release DO of 1.0 mg/L and warmed the release by 3.6 ^{o}C (Dortch and Wilhelms 1978). Although the pumping rate was nearly 50 percent of the discharge rate from the structure, additional improvement could have been possible with an improved design. In addition, these researchers stated that the application of this technique at other projects with higher discharges or deeper pools may not be as successful (Okatibbee Lake is approximately 9 m deep).

In 1980, a series of localized mixing tests were conducted on Pine Creek Lake and Lake Texoma in Oklahoma (Robinson 1981; Robinson, Garton, and Punnett 1982). These tests, using axial flow pumps with diameters of 1.22, 1.83, and 2.44 m, were designed to investigate pump performance with varying propeller diameter and reservoir release rates.

Conclusions from these tests indicated that improvement in release quality was feasible, but only for water quality constituents that display an increase with depth (excluding DO and temperature), and that increases in the pump propeller diameter increased the discharge ratio or dilution factor. However, a reduction in water quality was observed when the pumping rate exceeded the release rate. This may have been due to overpenetration of the pumped jet since the outlet level for these tests was at an intermediate level in the pool. The researchers indicated that an optimum ratio of pumped to release rate was observed for each series of tests.

Holland (1984) developed computational methods to predict the depth of penetration of a hydraulic jet as well as the entrained flow that crosses the thermocline. In addition, a method for designing a localized mixing system was developed (Holland 1983). Brown, Mobley, and Nubbe (1988) formulated a one-dimensional numerical model of a hydraulic jet in a withdrawal zone of the hydropower project at Douglas Dam. Field measurements of velocity versus depth for a large diameter plume from the surface water pump installation at Cherokee Dam are reported by Tyson and Mobley (1996). A two dimensional numerical model using PHOENICS code was developed to optimize the dissolved oxygen improvement for a given thermal stratification without having the pump plume disturbing bottom sediments (Hadjerioua 1996).

In 1987, a series of localized mixing tests were conducted at J. Percy Priest Reservoir in Tennessee (Price 1988, Sneed 1988). These tests were designed to investigate the feasibility of direct-drive mixers to improve release quality from the hydropower project. Results of these tests indicated that three pumps generating 45 cfs each improved the release DO by 1 mg/L (release discharge of 4,600 cfs). These tests also demonstrated that the location of the jet relative to the intake structure had a significant impact on the efficiency of the localized mixing application.

Similar tests were conducted at Douglas Reservoir by the Tennessee Valley Authority (Mobley and Harshbarger 1987). These tests, using three axial flow pumps, each with a propeller diameter of 4.5 m, indicated that the hydropower release DO could be improved by as much as 2.0 mg/L (Brown, Mobley, and Nubbe 1988). The results of these tests lead to further testing of commercially available mixing equipment for this application (Mobley, 1990), and eventually permanent full scale installations at TVA's Douglas Dam in 1994 (Mobley et al 1995), and Cherokee Dam in 1995 (Tyson and Mobley 1996) . The permanent installations at each dam cost approximately $2M and employ 9 surface water pumps to improve the release DO of 4 hydroturbine units by 1.5 to 3.0 mg/L depending on reservoir conditions. The surface water pumps at Douglas Dam have a 15 foot (4.5 m) diameter stainless steel impeller driven by a 40 HP electric motor. The surface water pumps at Cherokee Dam have composite impellers that are 16 feet in diameter and are driven by 75 HP electric motors. At each dam the pumps are designed to allow for up to 65 feet of headwater fluctuation. TVA has obtained good results from these installations and has found maintenance requirements for the electromechanical equipment and debris to be significant but reasonable compared with the other aeration alternatives used at each dam.

4.6.2.5 Summary

Localized mixing is a technique that is being successfully used to improve the release quality from reservoirs. This technique requires the use of a mechanical pump to jet surface water down into the hypolimnion with enough energy to penetrate to the center-line elevation of the intake port. A method for designing a localized mixing system was discussed, along with references for computational procedures. Applications of localized mixing have been tested in a number of locations with axial flow and direct-drive mixers and large systems have been permanently installed at 2 TVA dams since 1994. Several numerical techniques have been formulated which may assist in the design of a localized mixing system. Summary information on this technique is given in Table 4.6.3.

Table 4.6.3

Summary of Localized Mixing

Characteristic	Description
Targets	Improved water quality by mixing or diluting hypolimnetic waters with epilimnetic waters.
Mode of action	Surface pumps are used to force surface waters down into the hypolimnion and the withdrawal zone.
Effectiveness	Effective for stable stratification.
Longevity	Seasonal.
Negative features	Possible overpenetration of the jet may strike the bottom and cause resuspension of sediment and erosion.
	Temperature increase in releases associated with using warm surface water for DO improvement may be undesirable, especially for cold water fisheries in tailwater.
	Partial destratification of an area is possible.
	Significant maintenance is required for electromechanical equipment, flexible connections and debris control.
Costs	Moderate.
Applicability to reservoirs	Very applicable to reservoirs without selective withdrawal structures to increase the withdrawal of surface waters.

4.6.3 REFERENCES

Albertson, M. L., Dai, Y. B., Jensen, R. A., and Rouse, H. 1950. "Diffusion of Submerged Jets," *Transactions of American Society of Civil Engineers*, Vol 115, pp 639-697.

Bowen, L. 1981. "Some destratification studies in New South Wales," *Destratification of Lake and Reservoirs to Improve Water Quality*, Australian Government Publishing Service, Canberra, Australia, pp 771-778.

Brown, R. T., Mobley, M. H., and Nubbe, C. F. 1988. "Surface Water Pumps: Model-Prototype Calibration," *Proceedings, American Society of Civil Engineers International Symposium on Model-Prototype Correlation of Hydraulic Structures*, Colorado Springs, CO.

Burns, F. L. 1981. "Experiences in the design, installation and operation of destratification aerators," *Destratification of Lake and Reservoirs to Improve Water Quality*, Australian Government Publishing Service, Canberra, Australia, pp 709-725.

Croome, R. 1981. "Artificial destratification of two South Australian reservoirs," *Destratification of Lake and Reservoirs to Improve Water Quality*, Australian Government Publishing Service, Canberra, Australia, pp 737-760.

Davis, J. M. 1980 (Aug). "Destratification of Reservoirs - A Design Approach for Perforated Pipe Compressed Air Systems," *Water Services*, pp 497-504.

Davis, J. D., Holland, J. P., Schneider, M. L., and Wilhelms, S. C. 1987. "SELECT: A Numerical, One-Dimensional Model for Selective Withdrawal," Instruction Report E-87-2, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Dortch, M. S., and Wilhelms, S. C. 1978. "Enhancement of Releases from Stratified Impoundment by Localized Mixing, Okatibbee Lake, MS," Miscellaneous Paper H-78-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Garton, J. E., and Rice, C. E. 1976. "Improving the Quality of Water Releases from Reservoirs by Means of a Large Diameter Pump," Final Technical Report, Oklahoma Water Resources Research Institute, Stillwater, OK.

Hadjerioua, B., Mobley, M., Hauser, Gary E. and Brock, Gary. 1996. "Modeling of Surface Water Pumps in TVA Reservoirs", Proceedings of the ASCE 1996 North American Water and Environment Congress, Anaheim, California, June, 1996

Holland, J. P. 1983. "Localized Mixing to Enhance Reservoir Release Quality," Environmental and Water Quality Operational Studies Bulletin, Vol E83-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Holland, J. P. 1984. "Parametric Investigation of Localized Mixing in Reservoirs," Technical Report E-84-7, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Johnson, P. L. 1984. "Thoughts on selection and design of reservoir aeration devices," *Lake and Reservoir Management*, Proceedings of the Third Annual Conference of the North American Lake Management Society, US Environmental Protection Agency, Washington, DC, pp 537-541.

Kranenburg, C. 1979. "Lake destratification induced by local air injection," Report No. 79-2, Communications on Hydraulics, Department of Civil Engineering, Delft University of Technology, Netherlands.

Lorenzen, M., and Fast, A. 1977. "A Guide to aeration/circulation techniques for lake management," Report No. 600/3-77-004, Corvallis Environmental Research Laboratory, US Environmental Protection Agency, Corvallis, OR.

Mobley, Mark H. 1990. "Epilimnetic Dissolved Oxygen Content and Surface Water Pump Performance" , Proceedings: 2nd International Symposium on Gas Transfer at Water Surfaces Minneapolis, MN, Sept. 1990

Mobley, M. H., and Harshbarger, E. D. 1987. "Epilimnetic Pumps to Improve Reservoir Releases," in "Proceedings: CE Workshop on Reservoir Releases," Miscellaneous Paper E-87-3, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Mobley, M. H., W. C. Tyson, J. Webb, and W. G. Brock, 1995 “Surface Water Pumps to Improve Dissolved Oxygen Content of Hydropower Releases,” WaterPower '95, San Francisco, California, July 1995.

Moon, J. J., McLaughlin, D. K, and Moretti, P. M. 1979 (Sept). "Enhancement of Reservoir Release Water Quality by Localized Mixing --Hydraulic Model Investigation," Phase 1 Report, Contract No. DACW39-78-C-0045, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Price, R. E. 1988. "Evaluation of Release Improvement Techniques for J. Percy Priest Reservoir," Miscellaneous Paper HL-88-6, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Robinson, K. M., Garton, J. E., and Punnett, R. E. 1982. "Localized Destratification at Lake Texoma, " *Journal of the Environmental Engineering Division*, ASCE, Vol 108, pp 739-749.

Robinson, K. M. 1981. "Reservoir Release Water Quality Improvement by Localized Destratification," M.S. thesis, Oklahoma State University, Stillwater, OK.

Smith, J. 1981. "Canberra Destratification Pilot Study," *Destratification of Lake and Reservoirs to Improve Water Quality*, Australian Government Publishing Service, Canberra, Australia, pp 726-733.

Sneed, R. B. 1988. "Field Testing of Localized Mixing System - J. Percy Priest Lake, Tennessee," *Seminar Proceedings: Water Quality '88*,- US Army Corps of Engineers, Committee on Water Quality, Washington, DC.

Tyson, W. C, and M. H. Mobley. 1996. “Surface Water Pump Velocities at Cherokee Dam” TVA Report number WR28-2-12-105, May 1996

US Army Engineer District, Savannah. 1973 (Jun). "Allatoona Lake Destratification Equipment Test Report," prepared for US Environmental Protection Agency, Washington, DC, by USAED, Savannah, Savannah, GA.

4.7 IN-STRUCTURE TECHNIQUES: TURBINE VENTING, AUTO-VENTING TURBINE TECHNOLOGY, AND FORCED AIR

4.7.1 PROBLEM ADDRESSED

Low dissolved oxygen is a common problem with releases from hydropower projects during periods when the upstream reservoir is stratified. Most Francis turbines and some Kaplan turbines offer an opportunity to reaerate the release flow as it passes through the turbines. This can be accomplished at some sites by supplying air to openings in the turbine where the pressure is subatmospheric (turbine venting, auto-venting turbines), and at other sites by application of blowers or compressors (forced air systems).

4.7.2 TURBINE VENTING

4.7.2.1 Description

For many existing turbines, the vacuum breaker system can be modified or a separate venting system can be retrofitted to induce air into the discharge. These types of systems achieve, at most, about 30 percent reduction of the penstock DO deficit[1]. This generally means a maximum oxygen uptake of about 2.5 - 3.0 mg/l for a penstock DO of 1.5 mg/l and water temperature of 18^{o}C.

Advantages of this technique include (a) it is a passive system, often requiring only valves and piping to implement, and (b) installation and maintenance costs are relatively small compared to other DO enhancement alternatives. Two disadvantages of this system are (a) the absorption of nitrogen from aspirated air bubbles, resulting in total dissolved gas supersaturation, which may pose a downstream water quality problem, (b) a reduction in generating efficiency and capacity, and (c) the potential for increased noise in the powerhouse.

4.7.2.2 Theory

Turbine venting is the introduction of air, typically into the region of subatmospheric pressure downstream of the turbine blades. The pressures in this region are controlled by many factors, including the geometry of the turbine and draft tube, flow rate, tailwater elevation, and the elevation of the turbine. Therefore, turbine venting will not be applicable to all turbines. The magnitude of subatmospheric pressure and hydraulic characteristics of the air passageways determine the rate of air flow into the draft tube. Once the air is aspirated into the flow, the intense turbulence and transport

[1]DO deficit is the difference between the penstock oxygen concentration and the oxygen saturation concentration.

through submerged, high pressure regions of the draft tube and tailwater contribute to the oxygen transfer (Figure 4.7.1).

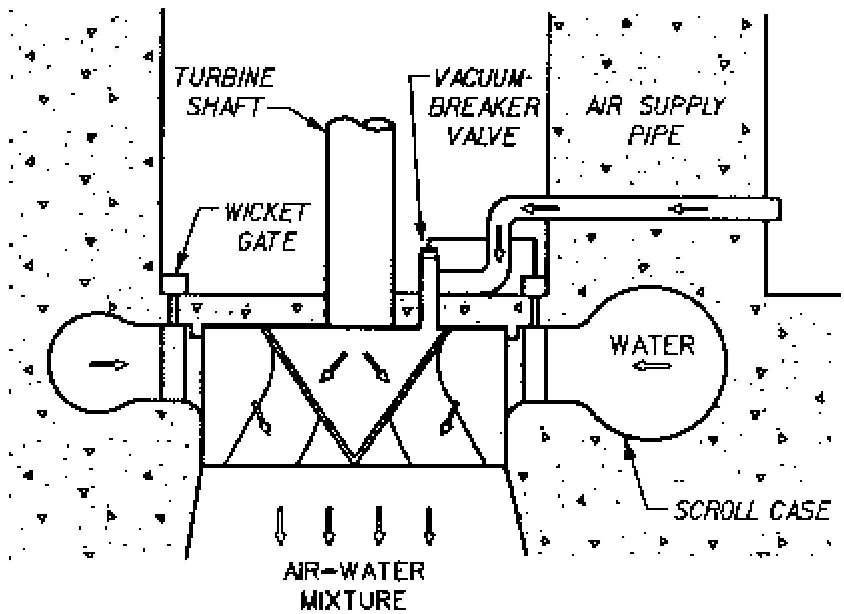


Figure 4.7.1 Vacuum Breaker Venting System

4.7.2.3 Design Methodology

Successful implementation of this technique requires a combination of both hydraulic analyses and field tests to determine the most effective method for introducing air into the turbine or draft tube. The following paragraphs describe a general methodology based on previous investigations and applications of turbine venting.

The analyses should begin with a description of existing hydrologic and water quality conditions. This can be accomplished by investigating the seasonal characteristics of in-reservoir temperature and DO and the resulting oxygen released without turbine venting. The observed water quality data can verify the application of a numerical model, such as SELECT (Davis et al. 1987), by comparing predicted and observed release quality. Once verified, the SELECT model can be used to estimate impacts of turbine venting by conducting simulations using the venting subroutine in the model. The venting subroutine assumes that a maximum of 30 percent of the upstream DO deficit can be satisfied by turbine venting.

If the results of the SELECT simulations indicate that an acceptable increase in release DO is predicted, field tests of an appropriate venting system are recommended. This testing should consider the specific turbine design and operating conditions to determine if venting of air is feasible and to identify potential locations for introducing air into the flow. This may involve modification of the vacuum-breaker system to allow more air to enter through the vacuum-breaker ports or the addition of another venting system. Such modifications usually include changes to the vacuum-breaker valve, which often must be maintained in an open position or replaced with an new operational system.

To increase subatmospheric pressures at aeration outlets, deflectors are often added to the turbine just upstream of the vacuum-breaker ports. These deflectors create a low-pressure region in their wake, thereby increasing air flow into the draft tube. Research by TVA has indicated that such "hub baffles" are very effective in extending the hydroturbine operating range over which air can be drawn into the water flow as shown in Figure 4.7.2 (Carter and Harshbarger, Sep 1997).

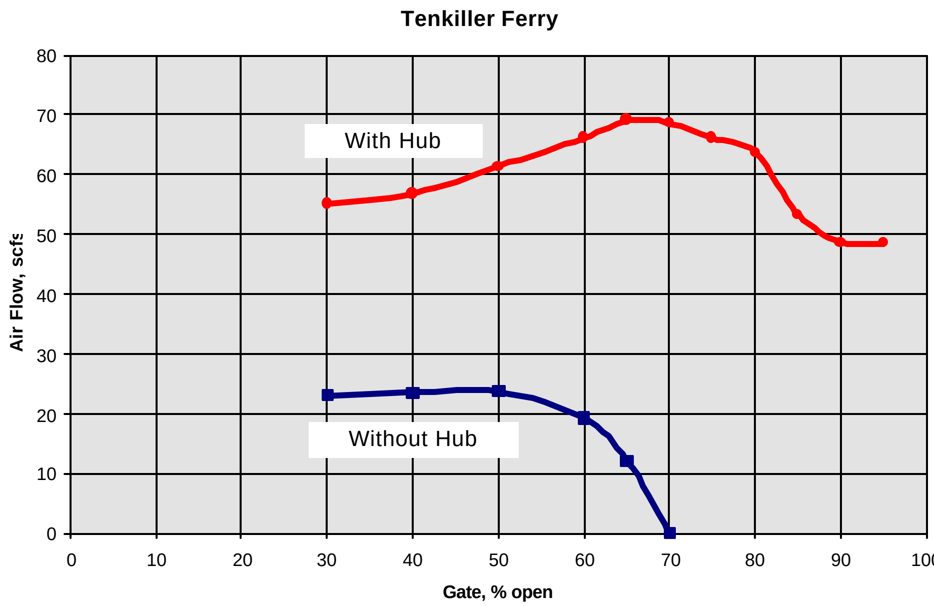


Figure 4.7.2 Effect of Hub Baffles on Air Flow

Modifications to the venting system should be tested to determine their effectiveness. In conjunction with these tests, turbine performance should be monitored because a loss in generation efficiency occurs during turbine venting (Almquist, Hopping and March, 1991, Wilhelms, Schneider, and Howington, 1987).

4.7.2.4 Applications

Several applications of turbine venting have been made at CE projects. The report by Wilhelms, Schneider, and Howington (1987) includes a comprehensive analysis of a turbine venting applications at CE projects. In tests on Clarks Hill Dam powerhouse (now J.S. Thurmund), hub baffles were added to the vacuum-breaker system to increase air flow. Turbine efficiency monitored during the tests, indicated an efficiency loss of approximately 2- to 3-percent was observed as a result of the venting. However, there was an increase in the release DO of approximately 2.5 mg/L or a reduction of the DO deficit of approximately 30 percent. A numerical model of turbine venting was developed from this work to predict the impacts of various air flow rates on release dissolved oxygen.

J. Percy Priest Reservoir on the Stones River in the Nashville District has operated with turbine venting for a number of years. At this project, the vacuum-breaker system is blocked open so that air can be entrained in the draft tube at all generation cycles. This results in a deficit reduction of approximately 27 percent, with an increase in the release DO of up to 2.0 mg/L (Price,1988).

Turbine venting has been applied at a number of hydroplants in the Alabama Power system. (Bohac et. al, 1983). At some of these sites, existing air piping used for tailwater suppression was used to supply air. At these sites, DO levels were raised about 0.5 to 1.0 mg/L and generation efficiency was decreased by up to 2%, depending upon the amount of air induced. At other sites, deflector baffles were added to the draft tube walls to cause local negative pressures for air induction. Air was supplied to openings just downstream from the deflectors by specially installed air supply piping. These systems were found capable of increasing the DO by 0.5 mg/L up to 1.8 mg/L, but required retrofitting piping systems and also reportedly caused about 2 % loss in generation efficiency.

Duke Power Company applied turbine venting at the Wylie Station using the existing vacuum breaker system (Bohac et. al., 1983). Significant aeration was obtained at reduced turbine loads, but at high turbine loads, there was no negative pressure to induce air.

A summary of the early turbine venting designs and hub baffles tested by TVA is included in Bohac et al. (1983). Since 1983, TVA has implemented turbine venting at 9 TVA hydropower projects (Carter 1995) and has recently assisted in the implementation of turbine venting at 5 CE projects such as Norfork, Tenkiller Ferry and Hartwell (Carter and Harshbarger, 1997; Carter and Harshbarger, 1998), and an investor owned utility project at Saluda (Kleinschmidt Associates, 1996). These turbine venting systems are used as a stand alone aeration solution or as a part of several aeration systems combined to meet downstream water quality targets.

TVA's successful implementation of turbine venting thus far has applied been exclusively to Francis units that exhibit some negative pressure under the headcover during operation. Site specific modifications are made to the turbine to increase the suction at the vacuum breaker outlet and to the supply piping to reduce pressure losses for air entering the unit. Together, these changes increase the air aspirated into the turbine and the amount of DO uptake. The most successful TVA installations provide 2 to 3 mg/L uptake and do not require limitations on the turbine gate operation. Turbine efficiency losses have been reported in the range of negligible to 1.5% with the majority of the losses occurring only during air entrainment. Turbine shaft deflections during air entrainment do not change significantly or tend to increase slightly at some units but have not been a problem. Cavitation damage to the TVA turbines has not increased significantly although wear patterns have sometimes been changed due to the air entrainment and hub baffles. Overall, TVA's experience with turbine venting has led to this being the immediate option of choice for aeration of hydropower releases. It is being used wherever feasible to avoid or reduce costs associated with other aeration alternatives.

4.7.2.5 Summary

Turbine venting is a technique successfully used at many hydropower projects to increase the DO in reservoir releases. This technique involves the introduction of air into regions of subatmospheric pressure downstream of the turbine blades. As this entrained air moves through the draft tube, the increase in the hydrostatic pressure increases the oxygen transfer efficiency, thereby increasing the DO uptake in the release.

A general approach to evaluation and design has been described which involves the use of a numerical model and prototype testing. Turbine venting has been successfully implemented by government agencies and investor owned utilities at many projects, with results that are very site specific. These turbine venting applications result in some loss in generation efficiency, dependent on airflow volume and location, and have generally experienced no problems with increased cavitation damage, shaft deflection, or vibration. Summary information is presented in Table 4.7.1.

4.7.3 AUTO-VENTING TURBINE TECHNOLOGY

4.7.3.1 Description

For turbines that are being replaced or upgraded, unique opportunities exist to optimize the turbine's hydraulic and aeration performance. Auto-venting turbine technology is simply turbine venting applied to a new unit where a "clean sheet" design approach can be used to optimize both turbine hydraulic performance and aeration. Although aeration considerations and design may delay the turbine unit upgrade schedule, utilizing an upgrade outage may provide opportunities for better aeration than would be possible or economical with retrofit systems.

4.7.3.2 Theory

For new turbines, a wide range of design factors, and consequently potential aeration alternatives exist to improve aeration performance. This is due to the flexibility in selecting the shape and position of key turbine components during the design and construction of new equipment. Although some limitations may be imposed by existing structures, new air passageway designs can be implemented more economically in the periphery of the water passageways while the turbine is removed, and air passageways can be designed into the turbine itself.

Table 4.7.1

Summary of Turbine Venting

Characteristic	Description
Targets	Low DO levels of the release waters commonly associated with hydropower generation during stratified periods and anoxic conditions.
Mode of action	Injection of air downstream of the turbine runner blades to increase the DO levels of the release waters.
Effectiveness	Up to 30 percent of the DO deficit may be satisfied by this technique.
Longevity	Based on the life of the turbine; years.
Negative features	Slight reduction in the generating efficiency and, consequently, the related revenues. Possible nitrogen supersaturation. Possible increase in the wear on the turbines
Costs	Low to Moderate depending on the required modifications to the turbines.
Applicability to reservoirs	Very applicable to hydropower projects.

4.7.3.3 Design Methodology

Numerical and physical models have been used to predict the aeration performance and hydraulic performance of various auto-venting alternatives. A detailed description of the current state of the art for auto-venting technology is included in a report prepared for the US Department of Energy Advanced Hydro Turbine Program (Franke et al., 1997). The basic requirement of the auto-venting design is to provide air supply passageways to subatmospheric pressures areas within the turbine. Numerical models used for turbine hydraulic performance design are frequently used to determine the location of these low pressure areas, exist or determine where they can be created. Other numerical models are used to predict the oxygen transfer to the water once the air is entrained into the water flow (Ventikos et al., 1998, Wilhelms et al., 1987, Buck et al., 1980) . Physical models have been used to evaluate the performance of various air passageway alternatives and choose optimal designs (March et al., 1991, March et al., 1992). Interpretation of physical model results requires scaling relationships such as those developed by Thompson and Gulliver (1997). Overall, the prediction of the performance of a hydroturbine with two-phase air/water flow is extremely complicated and site specific. However, better results are emerging as full-scale test data becomes available to enhance understanding of the turbine aeration problem.

4.7.3.4 Applications

TVA installed 2 auto-venting turbines at Norris Dam in 1995 and 1996. These units have been successful at supplying DO uptake values of up to 5.5 mg/L to support TVA's goals of 6 mg/L for the extremely popular cold water trout fishery downstream. Each turbine at Norris includes 4 air passageway designs as shown in Figure 4.7.3.

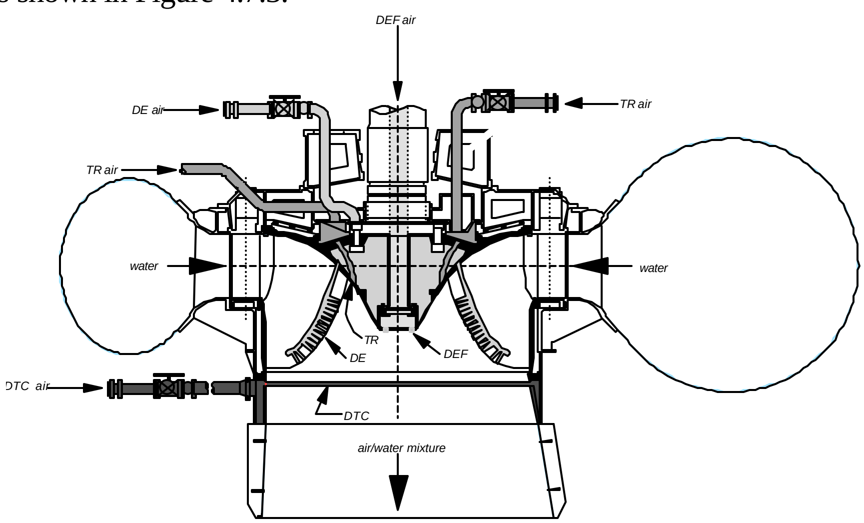


Figure 4.7.3 Aerating Turbine for Norris Hydro Project

These turbines have been tested extensively to evaluate the performance of the various aeration alternatives included in their design (Hopping et al., 1996, Hopping et al., 1997). A schedule of operation has been developed that optimizes the effectiveness of each aeration alternative in providing DO uptake with its effects on turbine efficiency at various operating conditions.

TVA is currently installing auto-venting turbines at Douglas Dam and has specified auto-venting requirements for a total of twenty six replacement units.

Other hydropower utilities are also making use of auto-venting technology. Duke Energy has installed new auto-venting turbines at their Wylie and Oxford Hydro Stations with preliminary results that are promising (Jablonski and Kirejczyk, 1998). Duke has ordered an additional auto-venting unit for Wateree. The Savannah and Mobile Districts are considering aeration requirements in specifications for new turbines at sites with low dissolved oxygen.

4.7.3.5 Summary

Auto-venting turbine technology is becoming an accepted means for economically improving the environmental performance of new turbines (Fisher et al., 1998). The design methodology is complex but becoming better understood as more installations are completed and tested. The successful application of the technology depends on the specific characteristics of the turbine and water passageways and may not be suited for all installations. Summary information is available in Table 4.7.2.

Table 4.7.2

Summary of Auto-Venting Turbine Technology

Characteristic	Description
Targets	Low DO levels of the release waters commonly associated with hydropower generation during stratified periods and anoxic conditions.
Mode of action	Aspiration of air into turbine water flow through passages specifically designed for aeration and implemented during turbine replacement to increase the DO levels of the release waters.
Effectiveness	Up to 50 percent of the DO deficit may be satisfied by this technique.
Longevity	Based on the life of the turbine; years.
Negative features	Is not applicable to all projects. Significant costs and time involved in analysis and design of aeration alternatives. Slight reduction in the generating efficiency during aeration and, consequently, the related revenues. Possible nitrogen supersaturation.
Costs	Low to Moderate depending on the complexity of the aeration alternatives.
Applicability to reservoirs	Applicable to hydropower projects that are being upgraded..

4.7.4 FORCED AIR

4.7.4.1 Description

For turbines that do not exhibit negative pressures under the headcover or do not induce sufficient air flow, a forced air system can be applied. Forced air refers to a system where a blower or air compressor is used to move the desired airflow into the turbine water flow passages.

4.7.4.2 Design Methodology

The design for forced air systems is straight forward compared to turbine venting and auto-venting since the air flow is not dependent on the subatmospheric pressure areas in the water flow. Many successful applications have been sized and designed based on full scale tests using portable air compressors connected to existing air passageways in the turbine. These tests are used to determine the effect of various air flows on the turbine, pressure requirements and oxygen transfer efficiency (Harshbarger, 1983).

Design considerations include; space restrictions in and around the powerhouse, availability of electric power, equipment reliability and total dissolved gas (TDG) limitations. The shear size and massive power requirements of forced air systems for large hydropower projects render some applications unwieldy or undesirable.

4.7.4.3 Applications

The TVA has forced air installations at Tims Ford Dam and Nottely Dam. Each of these projects have large air blowers on the main turbine and small air compressors on a small hydropower turbine used to provide minimum flows. Both projects are remotely operated, and compressor maintenance and operation has occasionally been a problem. Tims Ford is particularly susceptible to TDG supersaturation since the forced air system is supplying up to 3 mg/L of DO uptake. In fact, the tailwater TDG is routinely monitored and the aeration load is switched to oxygen diffuser systems on both large and small units as TDG limits are approached.

4.7.4.4 Summary

Forced air is a relatively straight forward aeration method that can often be designed based on full scale tests at the hydro project. Application may be limited due to TDG concerns or space and power requirements. Summary information is presented in Table 4.7.3. Aeration experience in retrofitting existing turbines is shown in Table 4.7.4.

Table 4.7.3

Summary of Forced Air Turbine Venting

Characteristic	Description
Targets	Low DO levels of the release waters commonly associated with hydropower generation during stratified periods and anoxic conditions.
Mode of action	Blowers or compressors are used to force air into turbine water flow through passages to increase the DO levels of the release waters.
Effectiveness	Large DO uptake values can be obtained by this technique but are limited by TDG supersaturation.
Longevity	Based on the life of the turbine; years.
Negative features	Maintenance requirements for rotating machinery, Power and space requirements in the powerhouse. Reduction in turbine generating efficiency during aeration and, consequently, the related revenues. Possible nitrogen supersaturation.
Costs	Moderate depending on the complexity and size of the forced air system.
Applicability to reservoirs	Applicable to hydropower projects that do not meet objectives with turbine venting.

4.7.5 REFERENCES

Almquist, C. W., P. N. Hopping, and P. A. March, "Energy Losses Due to Air Admission in Hydroturbines," *Proceedings of the ASCE National Hydraulic Engineering Conference,* Nashville, TN, August 1991.

Bohac, C. E., Boyd, J. W., Harshbarger, E. D., and Lewis, A. R. 1983. "Techniques for Reaeration of Hydropower Releases," Technical Report E-83-5, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Buck, C. L., D. E. Miller, and A. R. Sheppard, "Prediction of Oxygen Uptake Capabilities in Hydraulic Turbines Utilizing Draft Tube Aeration Systems," *Research Paper,* Alabama Power Company, Environmental and Research Services, 1980.

Carter, James C., and E. Dean Harshbarger, April 1997, “Turbine Venting for Existing Units at Norfork Hydro Plant”, TVA report number WR97-4-600-116, Engineering Laboratory, Norris, TN

Carter, James C., and E. Dean Harshbarger, September 1997, “Turbine Venting for Existing Units at Tenkiller Ferry Hydro”, TVA report number WR97-2-600-117, Engineering Laboratory, Norris, TN

Carter, James C., and E. Dean Harshbarger, January 1998, “Turbine Venting for Existing Units at Hartwell Dam, Units 1-4”, TVA report number WR98-4-600-118, Engineering Laboratory, Norris, TN

Carter, James C. 1995, “Recent Experience With Turbine Venting at TVA”, ASCE WATERPOWER '95, San Francisco, California, July1995.

Davis, J. E., Holland, J. P., Schneider, M. L., and Wilhelms, S. C. 1987. "SELECT: A Numerical, One-Dimensional Model for Selective Withdrawal," Instruction Report E-87-2, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Fisher, R. K., P. A. March, D. Mathur, F. Sotiropoulos, and G. F. Franke, “Innovative Environmental Technologies Brighten Hydro’s Future,” Keynote Address, Nineteenth Symposium of the IAHR Section on Hydraulic Machinery and Cavitation, Singapore, September 1998.

Franke, G. F., D. R. Webb, R. K. Fisher, D. Mathur, P. N. Hopping, P. A. March, M. Headrick, I. Laczo, Y. Ventikos, and F. Sotiropoulos, “Development of Environmentally Advanced Hydropower Turbine System Design Concepts,” York, PA: Voith Hydro, Inc., Report No. 2677-0141, U.S. Department of Energy Contract DE-AC07-96ID13382, July 1997.

Harshbarger, E. Dean, 1983, “Forced Air Turbine Venting Studies - January Through December 1982”, TVA report WR28-2-600-104, TVA Engineering Laboratory, Norris, TN.

Hopping, P. N., P. A. March, T. A. Brice, and J. M. Cybularz, 1996, “Plans for Testing and Evaluating the New Auto-Venting Turbines at TVA’s Norris Hydro Project,” *Proceedings, North American Water and Environment Congress,* ASCE.

Hopping, P., P. March, T. Brice, and J. Cybularz, “Update on Development of Auto-Venting Turbine Technology,” *Proceedings of WATERPOWER ´97,* August 1997.

Jablonski, Timothy A. and Juliusz Kirejczyk, 1998, Proceedings HydroVision 98, Exploring Our New Frontiers, July 28-31, 1998, Reno, Nevada. HCI Publications

Jones, R. K., and P. A. March, “Efficiency and Cavitation Effects of Hydroturbine Venting,” *Proceedings of the ASCE National Hydraulic Engineering Conference*, Nashville, TN, August 1991.

Kleinschmidt Associates, 1996, “South Carolina Electric & Gas Company Saluda Hydro Plant Turbine Aeration System Study” Phase III Report, Kleinschmidt Associates, Columbia, SC.

March, P. A., and R. K. Fisher, “TVA’s Auto-Venting Turbines Increase Downstream Aeration,” *Hydro Review*, September 1996.

March, P. A., T. A. Brice, M. H. Mobley, and J. L. Cybularz, “Turbines for Solving the DO Dilemma,” *Hydro Review*, March 1992, pp. 30-36.

March, P. A., J. Cybularz, and B. G. Ragsdale, “Model Tests for the Evaluation of Auto-Venting Hydroturbines,” *Proceedings of the ASCE National Hydraulic Engineering Conference*, Nashville, TN, August 1991.

Thompson, E. J. and J. S. Gulliver, 1997, “Oxygen Transfer Similitude for a Vented Hydroturbine,” *Journal of Hydraulic Engineering*, ASCE, in press.

March, P. A., and B. G. Ragsdale, “Rehabilitation to Achieve Multiple Objectives: Progress in the Development of Auto-Venting Hydroturbines,” *Proceedings of the EPRI Hydro Operation and Maintenance Workshop*, Boston, Massachusetts, September 1989.

March, P. A., J. Cybularz, and B. G. Ragsdale, “Model Tests for the Evaluation of Auto-Venting Hydroturbines,” *Proceedings of the ASCE National Hydraulic Engineering Conference*, Nashville, TN, August 1991.

March, P. A., T. A. Brice, M. H. Mobley, and J. L. Cybularz, “Turbines for Solving the DO Dilemma,” *Hydro Review*, March 1992, pp. 30-36.

Price, R. E. 1988. "Evaluation of Release Improvement Techniques for J. Percy Priest Reservoir," Miscellaneous Paper HL-88-6, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Ventikos, Y., F. Sotiropoulos, P. A. March, and P. N. Hopping, “A Lagrangian/Eulerian Method for Simulating Dissolved Oxygen Transfer in Autoventing Hydroturbines,” *Proceedings of the European Community on Computational Methods in Applied Sciences (ECCOMAS 98)*, Athens, Greece, September 1998.

Wilhelms, S. C., M. L. Schneider, and S. E. Howington, “Improvement of Hydropower Release Dissolved Oxygen with Turbine Venting,” *Technical Report E-87-3*, U.S. Army Corps of Engineers, Waterways Experiment Station, 1987

Wilhelms, S. C., Schneider, M. L., and Howington, S. E. 1987. "Improvement of Hydropower Release Dissolved Oxygen with Turbine Venting," Technical Report E-87-3, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Table 4.7.4. Aeration Experience in Retrofitting Existing Turbines

Project	Owner	Turbines			Aeration Features	DO Uptake (mg/L)
		Type	No.Units	Power (MW)		
American Falls	Idaho Power	*	*	*	forced aeration thru draft tube	2.0
Apalachia	TVA	Francis	2	39.5	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.0
Bagnell	Union Electric	Francis	8	25.0	vacuum breaker	*
Baldeney	*	*	*	*	forced aeration	*
Bankhead	Alabama Power	Kaplan	1	*	auto-venting thru air distribution conduit with draft tube deflectors	1.7
Bartletts Ferry	Georgia Power	*	*	*	auto-venting thru air distribution conduit with draft tube deflectors	*
Blanchard	Minnesota Power & Light	*	3	*	auto-venting thru air distribution conduit with draft tube deflectors	*
Boone	TVA	Francis	3	25.7	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.0
Box Canyon	*	Francis	*	*	vacuum breaker	*
Bull Shoals 1-4	USACE	Francis	4	38.8	auto-venting thru vacuum breaker with hub baffles and bypass conduit	*
Bull Shoals 5-8	USACE	Francis	4	41.8	auto-venting thru vacuum breaker with hub baffles and bypass conduit	*
Cascade	Idaho Power	Kaplan	2	6.4	forced aeration thru draft tube	*
Cherokee	TVA	Francis	4	30.9	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.5
Clarks Hill	USACE	*	*	*	auto-venting thru draft tube air distribution conduit	1.5
Conowingo	Susquehanna Power	*	*	*	auto-venting thru vacuum breaker	*
Coyote	*	Francis	*	*	auto-venting thru vacuum breaker	*
Deer Creek	Bureau of Reclamation	Francis	2	2.5	auto-venting thru vacuum breaker and snorkel tube	1.2
Douglas	TVA	Francis	4	26.5	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.0
Fontana	TVA	Francis	3	68.2	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.5
Hiwassee	TVA	Francis	1	59.7	auto-venting thru vacuum breaker with hub baffles and bypass conduit	1.0
Holt	Alabama Power	*	1	*	*	*
Kaw	KAMO Electric Co-op.	*	*	*	forced aeration thru draft tube	*
Kimberly	*	Propeller	4	0.9	*	*
Logan Martin	Alabama Power	Propeller	3	42.0	auto-venting thru air distribution conduit with draft tube deflectors	0.5
Martin	Alabama Power		4	*	*	*
Mitchell	Alabama Power	Propeller	4	50.0	auto-venting thru air distribution conduit with draft tube deflectors	1.0
Monticello	*	Francis	*	*	vacuum breaker	*
Norfork	USACE	Francis	2	31.3	auto-venting thru vacuum breaker with hub baffles and bypass conduit	*
Norris**	TVA	Francis	2	64.7	auto-venting thru vacuum breaker with hub baffles and bypass conduit	5.5
Pixley	Flambeau Paper	Francis	2	0.6	auto-venting thru draft tube air distribution conduit	1.0
Poppenweiler	*	Kaplan	2	1.8	auto-venting thru draft tube air distribution conduit	*

Project	Owner	Turbines			Aeration Features	DO Uptake (mg/L)
		Type	No.Units	Power (MW)		
Rapide Croche	*	Propeller	1	0.8	auto-venting thru vacuum breaker	1.5
Rat Rapids	*	Francis	4	0.32	auto-venting thru vacuum breaker	0.8
R.L. Harris	Alabama Power	Kaplan	2	67.5	auto-venting thru air distribution conduit with draft tube deflectors	2.0
Rothschild	*	Francis	6	0.8	auto-venting thru vacuum breaker	1.5
Safe Harbor	Pennsylvania Power & Light	*	*	*	draft tube	*
Shepaug	Connecticut Light & Power	Kaplan	1	42.0	forced aeration thru draft tube	2.0
South Holston	TVA	Francis	1	36.2	auto-venting thru vacuum breaker and bypass conduit	2.0
Tenkiller Ferry	USACE	Francis	2	28.3	auto-venting thru vacuum breaker with hub baffles and bypass conduit	*
Tims Ford**	TVA	Mixed Flow	1	*	forced aeration thru vacuum breaker	*
Watauga	TVA	Francis	2	25.7	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.0
Winfield	American Electric Power Service	Kaplan	3	6.15	auto-venting thru vacuum breaker	3.0
Wyle	Duke Power	Francis	4	15.0	auto-venting thru vacuum breaker	0 - 2
Youghiogheny	*	*	1	*	forced aeration thru draft tube	2-5

*Missing Information
**Original Turbines

4.8 TAILWATER TECHNIQUES

4.8.1 MINIMUM FLOW

4.8.1.1 Description

Sustaining minimum flows in tailwater reaches provides 1) continuous wetted channel area to support benthic organisms; 2) reduced thermal shock for aquatic biota; 3) improved flushing of otherwise stagnant pools; and in certain cases, 4) improved water velocities and depths for fish spawning. This description provides an overview of potential mitigative technologies, including turbine pulsing, small turbine units, reregulating weirs, spilling/sluicing, and mobile systems.

Turbine Pulsing. Turbine pulsing is an operational method that normally results in the lowest power losses of any of the minimum flow options, and with essentially no capital investment. Turbine pulsing is the operation of one turbine at a hydropower dam for a short duration at a regular time interval (e.g., 0.5 hr generation every 4 hrs). Pulsing is performed to provide minimum flow during times when generation is not otherwise scheduled, such as at night, on weekends, or during filling of the reservoirs. Pulses typically dampen with a few miles downstream of the powerhouse, depending on stream hydraulic parameters like width, roughness, and slope. Wear and tear on switching mechanisms creates the primary O&M expense for turbines used for pulsing, but this can be minimized in multi-unit plants by rotating service across several turbines.

Small Turbine Units. A small turbine unit addition to provide minimum flow is a capital intensive, moderate O&M option. The advantage of using a small turbine unit for minimum flow is that flow is continuous at the release, and power is being generated while achieving the minimum flow target. An auxiliary mini-penstock is tapped into the most convenient powerhouse feature, such as the main penstock, sluiceway, surge tank, or turbine bypass piping. The mini-penstock is typically three to four feet in diameter and routed out of the powerhouse to supply the small generating unit. The small unit can either be attached to the powerhouse exterior or located on the bank downstream of the powerhouse.

Reregulation Weirs. Weirs designed for minimum flow are reliable, capital intensive, low-maintenance structures. Reregulation weirs less expensive than aeration weirs. A weir can reduce effective turbine head by increasing tailrace depths if the weir is not located sufficiently far downstream from the powerhouse. Overflow weirs, if not properly designed, can create downstream recirculation patterns that can pose a safety hazard to unwary river users. Safety is as much a part of weir design as the minimum flow.

Weirs typically regulate minimum flow during periods of non-generation via slow drainage of the weir pool through low-level pipes. These pipes can be controlled by self-actuating float-valves to maintain constant flow as the weir pool fluctuates (Hadjerioua, et. al. 1992 and 1997). The weir pools are typically recharged with a turbine release every 12 to 48 hours if not otherwise generating, depending on site characteristics. These weirs typically are not large enough to attenuate peaks during normal turbine operation.

Spilling/Sluicing/Siphons. In most cases it is economically desirable to pass all water for minimum flow through a project's main turbines. In certain instances where this is not possible (i.e., turbines out of service, or other options such as pulsing are not feasible), then spilling, sluicing, or siphoning the water are potential options. These systems usually require capital outlay to upgrade equipment sufficiently to handle cavitation and vibration issues.

Mobile Systems. If a minimum flow system relies solely on a single main turbine, then the minimum flow will not be sustainable during main unit outages. This can occur if a small unit's mini-penstock taps into main unit hardware, or if pulsing is used at a project that has only one turbine. To augment minimum flow during such outages, TVA has developed a mobile system (Schulte and Harshbarger, 1997) that can be deployed to pump or siphon reservoir water over the top of the spillway. The system consists of electric pumps mounted on rafts, and hoses to carry pumped water to the dam's spillway. The equipment is stored on trailers, in anticipation of a rapid response that may be needed to maintain habitat during an outage.

4.8.1.2 Design Considerations

Suitability of certain mitigative techniques varies by project. This section will offer design considerations based on experiences with these systems.

Turbine Pulsing. Not all tailwaters are good candidates for turbine pulsing (Hauser, 1989). For hydro projects with large turbines discharging into streams having relatively small natural flows, turbine pulsing cannot feasibly provide sustained flows small enough to be within the target range.

In the immediate tailrace, turbine pulsing creates flow fluctuations between the leakage flow rate and one turbine discharge. Shortly downstream, however, the fluctuations dampen such that the minimum and maximum flows approach a temporal mean. TVA has successfully pulsed certain hydroprojects in the range of 30 to 60 minutes on intervals ranging from 2 to 6 hours for a variety of downstream objectives. Figure 4.8.1 shows minimum, mean, and maximum flow profiles below Douglas Dam for a pulsing pattern of 0.5 hr of one turbine (4000 cfs) every 4 hrs. Use of pulsing at Douglas Dam to provide 550 cfs cost TVA about one-fourth that of a sluicing option that would have provided only 300 cfs continuous flow.

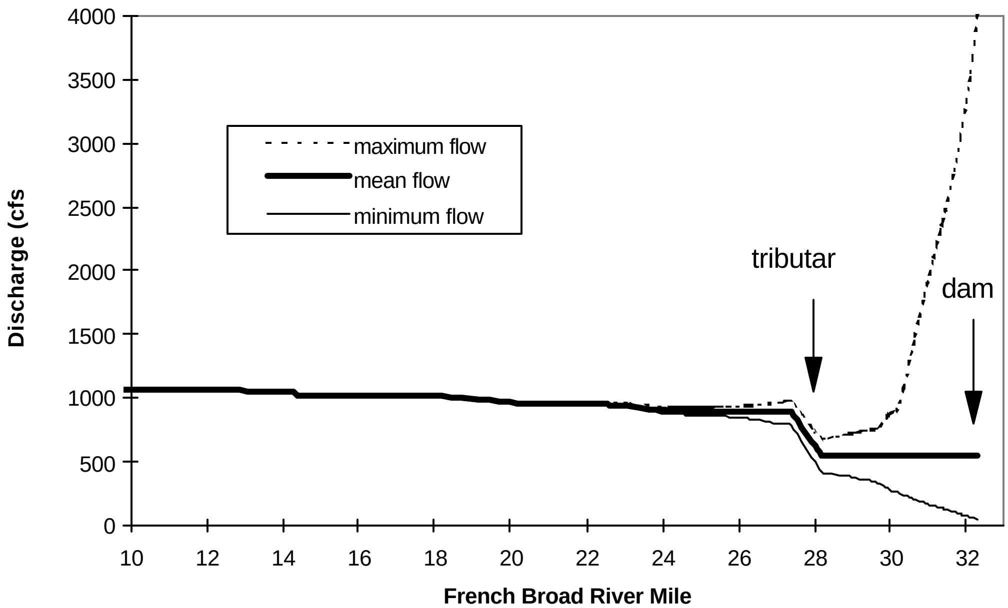


Figure 4.8.1 Discharge below Douglas Dam with Turbine Pulsing

Small Turbine Units. Small turbines may be less efficient than the larger turbines used for hydrogeneration, especially if salvaged hardware is used. The advantage of using small units to provide a minimum flow is that flow is constant near the release, and power is being generated while achieving the minimum flow target. The operating efficiency of the salvaged hardware installed at TVA's Tims Ford and Nottely applications is about 60%, whereas, machinery manufactured specifically for small unit installation at Blue Ridge has an efficiency of about 90%. Addition of a small unit for minimum flow requires additional consideration for aeration the small unit release.

Reregulation Weirs. TVA's Norris weir shape was developed in physical modeling. The surface of the weir is porous, and only the upstream face is impervious. Thus, water flows over the impervious face, then over and through the downstream rockfill portion of the weir, dissipating energy before the flow meets the weir tailwater. The gradual step-down shape from headwater to tailwater eliminates a sudden plunge into the weir tailwater, avoiding the dangerous recirculation historically associated with low-head weirs. Design considerations for locating and sizing reregulation weirs are similar to that described in aerating weirs used for dual aeration/minimum flow objectives in other sections of this document.

Spilling/sluicing. Water temperature is a consideration with this option when downstream fisheries are coldwater fisheries that have arisen from cold turbine releases. Another consideration is that cavitation potential can be high for low level sluices when minimum flows are discharged at high head through partial gate openings. In these cases, a special valve or sluice gate is required to pass the flow safely without equipment damage.

4.8.1.3 Applications

Turbine Pulsing. TVA began providing minimum flows via turbine pulsing below 8 dams (Appalachia, Boone, Cherokee, Douglas, Fontana, Fort Patrick Henry, Ocoee 1, and Wilbur) in 1991. Minimum flows provided are typically in the range from 0.5 to 1.5 the natural summertime 7Q10 at the site.

Small Turbine Units. TVA has small units at Tims Ford, Nottely, and Blue Ridge dams, completed 1986 and 1993. The small unit at Tims Ford is attached to the powerhouse exterior. The 91.44 cm (36 inch) diameter mini-penstock is tapped into the sluice way adjacent to the powerhouse. The small unit provides 2.12 m^3/s (75 cfs) to the downstream channel and generates 500 kW. The turbine is a salvaged pump from a nuclear plant application. Installing the pump backwards allows it to act like a turbine and generate electricity.

The small unit at Nottely is located on the right bank, about 45.8 m (50 yards) downstream of the powerhouse. The 36-inch diameter mini-penstock is flanged to the housing where an old regulating sleeve valve used to be. The small unit provides 1.42 m^3/s (50 cfs) to the tailwater reach and generates 400 kW. The hydropower machinery is also a backward-running pump obtained from a nuclear plant application.

The small unit at Blue Ridge uses a 121.9 cm (48-inch) diameter mini-penstock was attached to the surge tank of the main unit and routed 45.8 m (50 yards) downstream to the small unit turbine. The small unit provides 3.26 m^3/s (115 cfs) to the tailwater reach and generates 600 kW.

Reregulation Weirs. TVA has been developing, testing, designing, and constructing innovative weirs to provide minimum flows since 1983. The first TVA weir was a unique experimental flow reregulation weir in the Clinch river below Norris Dam. The original weir was constructed of galvanized steel gabion baskets. It was 1.52 m (5 ft) high, 6.4 m (21 ft) in upstream to downstream dimension, and 128.9 m (423 ft) long, and had 54 30.48 cm (12-inch) diameter steel pipes passing through it for discharge control (Shane, 1985). The weir deck had to be paved with concrete in March 1987 because of excessive deterioration of the gabion baskets. This modification resulted in the emergence of a recirculating flow pattern in the tailwater of the weir when only one turbine at Norris was discharging. The recirculation was eliminated by addition of a rock filled wood crib structure to the downstream end of the weir, which allowed water to flow through the structure as it did before the deck was paved. The through flow eliminated any recirculation (Loiseau, 1987).

The experimental weir structure was replaced with a permanent timber crib design in 1995. The new weir is a stepped timber crib filled with loose rock resting on a concrete pad and lined with tongue-and-groove pressure-treated timbers along its upstream face to make it impermeable. The concrete pad serves as a leveling pad for the structure, making it easier to maintain the weir elevations across the channel during construction and eliminating shifts in the structure due to changes in the channel bottom during operation.

The TVA South Holston labyrinth weir, completed in December 1991, was constructed 1.92 km (1.2 mi.) downstream from South Holston dam and is designed to sustain a minimum flow of 2.55 m^3/s (90 cfs) between generating periods and to increase river DO content to at least 6 mg/L. It is a long, W-shaped structure that creates a zigzag waterfall when the weir is overtopped during turbine operations. The weir can sustain the target minimum flow for up to 18 hours.. The South Holston labyrinth weir has been meeting or exceeding aeration and minimum flow objectives with minimal operation and maintenance.

The TVA Chatuge infuser weir, completed in 1992, provides a sustained minimum flow of 1.7 m^3/s (60 cfs) during non-generation periods via a 12-hour pulsing interval established at Chatuge hydro plant to maintain a weir pool. Six low-level pipes penetrating the weir to slowly release the stored water. Flow through two of these pipes is controlled by an orifice plate attached to the downstream end of the pipe, and the remaining four are controlled by self-actuating float valves.

Mobile Systems. TVA's emergency minimum flow system is used as a regional backup to minimum flow systems for single-turbine projects at Blue Ridge, Chatuge, and Nottely (Schulte and Harshbarger, 1997). These systems include lifts of from 18 ft at Chatuge to 65 ft at Blue Ridge, and they provide flows of 30 cfs at each location.

4.8.1.4 Summary

TVA has successfully supplemented minimum flows at over a dozen hydroplants by using turbine pulsing, small generating units, or reregulation weirs. Because each tailwater is different, there is no preferred method for all hydroplants. Each tailwater requires study to determine the optimum method, depending on the desired aquatic restoration goals for the project, the downstream uses, and the characteristics of the hydroplant and the tailwater.

With the exception of turbine pulsing, augmenting minimum flow requires substantial capital expenditures, but operational costs are low. Achieving a suitable yet cost-effective solution for all stakeholders can foster goodwill between the hydropower producer and the public, state agencies, and special interest groups.

4.8.2 TAILWATER TECHNIQUES: AERATING WEIRS

4.8.2.1 Description

Advantages of Weirs. The primary advantages of weirs used for aeration is that they are passive devices that provide excellent aeration, high reliability, with minimal operation and maintenance. Weirs can also meet a second environmental objective of maintaining minimum flow (wetted area) in the channel by releasing stored water slowly between generating periods. Weirs designed to aerate maximum turbine discharges will usually exceed aeration targets at lower turbine discharges. Unless they are dewatered, weirs will add oxygen in seasons when DO is not needed. Thus, weirs may produce a significant increment of aeration in excess of the year-round DO target, in contrast to systems throttled to just meet a flat DO target during the low DO season. Depending on the target DO levels, this may be bioenergetically significant for downstream fisheries. Unlike hydroplant aeration methods, weirs are not likely to affect cavitation or turbine efficiency. Potential power impacts from aerating weirs can be lost revenue during outages required for weir construction, and capacity losses from reduced turbine heads if the weir is constructed close to the hydroplant. Weirs are attractive to the public.

Disadvantages of Weirs. The primary disadvantage of weirs is that they can represent a significant capital investment, especially in deep river situations. Aerating weirs normally require a head differential of 1 to 2 m (3 to 7 ft) at all turbine discharges; thus they impound a pool of substantial depth upstream of the weir. Depending on proximity to the upstream dam and the stream gradient, this impoundment can reduce the effective head on the turbine and increase inundation of adjacent property along the weir pool during floods. Another disadvantage is that the weir pool is not aerated, leaving a portion of the tailwater without DO improvement. A weir increases the depth and residence time of water in the weir pool reach, increasing oxygen depletion there from organic materials in the sediments and water column and from chemical species released from the upstream reservoir. The public attraction to weirs as a recreational area can become undesirable if safety concerns are significant. Weirs may also create a barrier to migration of fish and to human navigation. However, for hydroplant aeration, this will not be nearly as significant as the barrier presented by the hydropower dam itself, except for the reach between the weir and upstream dam.

Weir Types. Considerable progress has been made in aerating weirs since the 1990 EPRI DO guidance (EPRI 1990). Figure 4.8.2 illustrates plan and elevation views of three basic types of aerating weirs that employ free-fall nappes: linear, labyrinth, and infuser.

The linear weir (Figure 4.8.2a) has been found in the literature (Gameson 1957, Avery and Novak 1978, Nakasone 1987) to be a good aerator at the lower specific discharges of 0.05 to 0.2 m^2/s (0.5 to 2 ft^2/s), where specific discharge is defined as flow per unit length of weir crest. At higher discharges, hydraulic recirculation downstream of the weir intensifies to a hazardous level. Aeration also reduces at high specific discharge.

The labyrinth (Hauser 1995, 1996) assumes a "W" shape in plan view to fit within the channel, as shown in Figure 4.8.2b. By increasing the crest length, the labyrinth weir reduces specific discharge, thereby improving both aeration and safety of the labyrinth compared to a conventional linear weir (Hauser 1991). The labyrinth weir's efficient head-discharge relationship allows it to pass the same flow at a reduced head compared to a linear weir spanning the same river width. Labyrinths can therefore be constructed closer to an upstream dam with less impact on turbine head.

The infuser weir (Figure 4.8.2c) was developed to achieve aeration equivalent to a labyrinth weir with a compact design that requires less space in the channel. Like the labyrinth, the infuser (Hauser and Brock 1994, Rizk and Hauser 1993) uses a long length of waterfall to achieve a high total nappe perimeter to flow ratio. Unlike the labyrinth, however, the infuser splits the flow into a tight series of transverse water curtains that impinge on a plunge pool beneath the infuser deck. Grating over the deck openings creates turbulent irregular-shaped waterfalls that further increase the nappe perimeter to flow ratio. Hydraulically, the infuser behaves like a broad-crest weir. Turbulent regions beneath the infuser deck are rendered off limits to people using a barrier cage at the downstream face of the infuser deck. Clogging of the infuser deck requires somewhat more routine maintenance than conventional or labyrinth weirs. The infuser weir concept has been patented by TVA, and is protected under U.S. Letters Patent Nos. 5,462,657 and 5,514,285. TVA can license this technology on a non-exclusive basis to interested organizations for their particular applications.

4.8.2.1 Theory

The hydraulics of labyrinth weirs are quite complex, but considerable literature (Hay and Taylor 1970, Indlekofer and Rouve 1975, Lux 1993), Tullis, et al (1993) exists because these weirs are used frequently for spillway applications.

The primary parameters influencing weir aeration efficiency are 1) drop height, defined as headwater minus tailwater across the weir; 2) specific discharge, defined as flow per unit length of the waterfall(s); 3) roughness of the waterfall perimeter as it impinges on the plunge pool; 4) plunge pool geometry. Weir aeration occurs along the surface and underside of the falling nappe, across the plunge pool surface, and across the surfaces of bubbles submerged in the plunge pool. Most of the total aeration at a weir occurs in the submerged bubble region. Air is transported along the boundaries of the free-falling nappe and introduced into the plunge pool. In the submerged bubble region, most of oxygen transfer occurs in the descending phase, as air introduced by nappe impingement is sheared into small bubbles and plunged downward toward the channel bed. Transfer also occurs as the bubbles ascend, but this transfer is less than during descent due to bubble coalescence, reduced turbulence, and buoyant separation of bubbles from the fluid jet containing lowest DO. A rougher surfaced nappe will typically have more perimeter per flow at impingement, improving its aeration efficiency. Aeration efficiency is lowest at high specific discharges (high flow to perimeter ratio) and at very low specific discharges (nappe impinges as a spray or mist); thus, an optimal specific discharge exists for various weir types

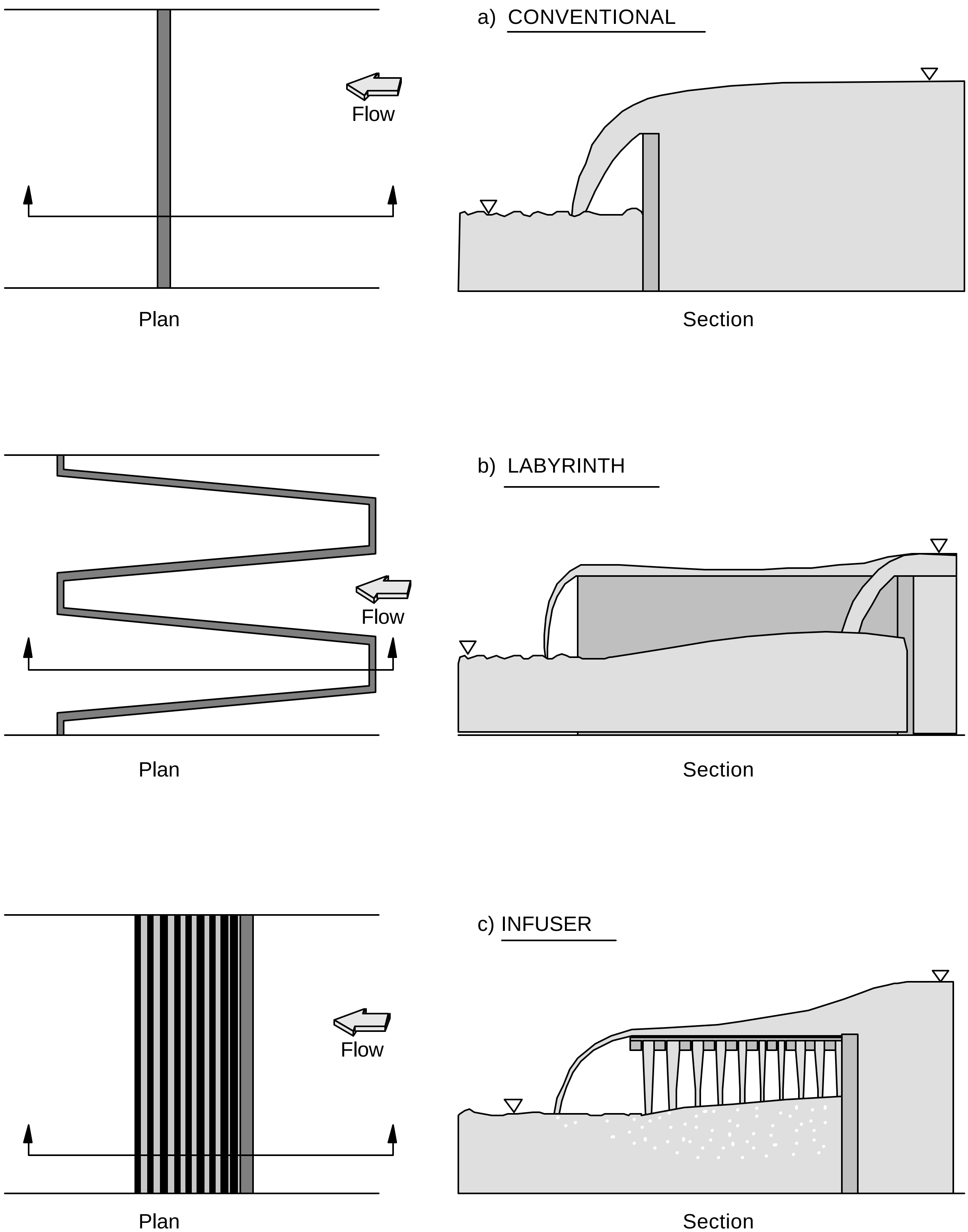


Figure 4.8.2 Aerating Weir Types

and plunge pool geometries. Obstructions in the plunge pool can dramatically alter flow patterns and affect aeration.

4.8.2.3 Design and Construction Considerations

Location. The weir should be located close to the upstream dam to maximize the length of improved tailwater but far enough to minimize undesirable reduction in the effective turbine head. Combining the aeration and minimum flow requires a location far enough downstream of the dam to provide adequate storage behind the weir to sustain a minimum flow between generating periods. Suitable foundation material for weir location is often found at locations of rock outcropping, where dewatering requirements during construction can be minimized. A labyrinth can require ten times the flat channel area as an infuser, making site selection more difficult where the exposed rock is irregular or not level. For efficient aeration, a sufficiently deep plunge pool must be available at all turbine flows, and it is often more economical to achieve the proper depth naturally via site selection rather than artificially constructing a plunge pool. Sites should avoid sediment or pollution sources in the weir pool.

Materials. Concrete is considered more durable and has a longer life expectancy than wood but is more costly, due largely to forming costs. Once constructed, concrete weirs are not as easily changed as a weir made of wood and concrete. TVA's South Holston weir, for example, uses tongue-and-groove pressure-treated timber walls supported in a stoplog arrangement by slotted concrete piers. The piers were constructed with reusable slip forms, and the wall sections were designed for easy replacement, should the need arise.

Weirs in Combination with other Aeration Systems. When used in combination with other aeration systems, weirs are more suitable as base aeration systems rather than topping systems, because aeration potential is fixed at construction by the water level difference across the weir. When used in conjunction with other technologies, it may be advantageous to pair a weir with a topping system rather than with another base system. Attempting to combine multiple aeration systems, such as a weir and turbine venting, can reveal site-specific incompatibilities. The backwater induced by the weir may adversely impact the aeration performance of the venting, which relies on a high turbine elevation relative to tailwater elevation to develop low pressures for natural aspiration. Also, DO depletion in a deep weir pool could act to negate part of the DO improvement added by an upstream method.

4.8.2.4 Applications

Although much literature has been published on the aeration characteristics of conventional weirs, few existing weirs were designed specifically for aeration. The Table 4.8.1 is a list of existing weirs designed specifically for aeration.

Table 4.8.1. Prototype Aerating Weirs				
Owner	Dam	Weir Type	Materials	Purpose
GBRA[1]	Canyon	labyrinth	concrete wall	DO
TVA[2]	S. Holston	labyrinth	timbers, concrete piers	DO, min. flow
TVA[2]	Chatuge	infuser	timber crib, concrete piers	DO, min. flow
Pennelec	Deep Creek	labyrinth	concrete wall	DO
GP[3]	Lloyd Shoals	labyrinth	steel sheet	DO
EDF[4]	Petit Saut	cascade labyrinth	steel frame, concrete wall	DO, degas methane

[1]Guadalupe-Blanco River Authority
[2]Tennessee Valley Authority
[3]Georgia Power
[4]Electricite de France (weir in French Guiana)

4.8.2.5 Summary

Weirs are reliable, effective, low maintenance aerators that can meet both aeration and minimum flow environmental objectives. Weirs represent a significant capital cost, but weirs can often be the more cost effective option when flows and capacity factors are high, or when being considered for both aeration and minimum flow objectives.

4.8.3 REFERENCES

Avery, S., and P. Novak. 1978 (Nov). "Oxygen Transfer at Hydraulic Structures;" *J. ASCE Hydraulics Division*.

Electric Power Research Institute. 1990 (Nov). "Assessment and Guide for Meeting Dissolved Oxygen Water Quality Standards for Hydroelectric Plant Discharges;" GS-7001; Research Project 2694-8.

Gameson, A. L. H. 1957. "Weirs and the Aeration of Rivers;" *J. Institute of Water Engineers*, 11:477-490.

Hadjerioua, Boualem; T. A. Rizk; E. M. Laursen; G. E. Hauser. 1994 (Mar). "Regulation of Flow Downstream of Weirs"; *J. ASCE Hydraulics Division*, Vol. 120, No. 3.

Hadjerioua, Bo, W. D. Proctor, and G. E. Hauser. 1997 (Apr). "Valve and Trash Rack Design - Center Hill Reregulation Weir," TVA Engineering Laboratory, Norris, TN, WR97-2-590-172.

Hauser, G. 1991. (May). "Full-Scale Physical Modeling of Plunge Pool Hydraulics Downstream of a Vertical Weir;" TVA Engineering Laboratory; WR28-1-590-153; Norris, TN.

Hauser, Gary E. 1991. (Dec). “Mathematical Modeling of a Rock Regulating Structure for Enhancement of Norris Reservoir Releases,” TVA Engineering Laboratory, Report No. WR28-1-2-109, Norris, TN.

Hauser, Gary E. 1989. (Sep). “Turbine Pulsing for Minimum Flow Maintenance Downstream from Tributary Projects, Technical Note,” TVA Engineering Laboratory, Report No. WR28-2-590-147, Norris, TN.

Hauser, G. E., and W. G. Brock. 1994 (Aug). "The Aerating Infuser for Increasing Dissolved Oxygen Concentrations"; HydroVision '94 and HydroReview magazine; Phoenix, Arizona.

Hauser, G. E., and Doug I. Morris. 1995 (Jul). “High Performance Aerating Weirs for Dissolved Oxygen Improvement”; *Proceedings WaterPower 95*.

Hauser, G. E., and Doug I. Morris. 1996 (Dec). “Aerating Weirs Design”; EPRI TR-103947; prepared for EPRI by Tennessee Valley Authority Engineering Laboratory.

Hay, N., and G. Taylor. 1970 (Nov). "Performance and Design of Labyrinth Weirs;” *J. ASCE Hydraulics Division*.

Indlekofer, H., and G. Rouve. 1975 (Mar). "Discharge Over Polygonal Weirs;" *J. ASCE Hydraulics Division*; 101(HY3):385-401.

Loiseau, Paul B. 1987 (Dec). “Physical Model Studies and Prototype Field Evaluation of Modifications to the Clinch River Flow Reregulation Weir,” TVA Engineering Laboratory, Report No. WR28-2-590-139, Norris, TN.

Lux, F. 1993 (Aug). "Design Methodologies for Labyrinth Weirs;” *Proceedings ASCE WaterPower 1993*; Nashville, TN.

Montgomery, C. R. et. al. 1997. **“**TVA Experience with Minimum Flow Techniques at Hydropower Dams”; TVA Engineering Lab; presented at WaterPower 97; Atlanta.

Nakasone, H. 1987 (Feb). "Study of Aeration at Weirs and Cascades", *J. ASCE Environmental Engineering*.

Rizk, T. A., and G. E. Hauser. 1993 (Jun). “Chatuge Aerating Infuser Physical Model Study”; WR28-1-17-101; TVA Engineering Laboratory; Norris, TN.

Schulte, D. W. and E. D. Harshbarger. 1997 (Aug). “Emergency Minimum Flow System for Dams”; TVA Resource Group; presented at Water Power 97; Atlanta, Georgia.

Shane, Richard M. 1985 (Jan). "Experimental Clinch River Flow Reregulation Weir Field Evaluation Interim Report", TVA Engineering Laboratory, Report No. WR28-4-590-118, Norris, TN.

Tullis, J. Paul, Amanian, N., Wasdron, D. R. 1993. "Design Procedure for Labyrinth Spillways"; Technical Review; Two-part series; *Association of State Dam Safety Officials (ASDSO) Newsletter*.

4.9 ALTERNATE MANAGEMENT OPPORTUNITIES

INTRODUCTION

Additional management opportunities exist in methods oriented towards nutrient management or chemical remediation. These techniques often utilize a more biologically oriented approach than traditional "engineering" approaches previously described. A partial list of these techniques is provided in table 4.9.1. These methods are discussed in more detail in Cooke and Kennedy (1989). Excerpts from Cooke and Kennedy (1989) are included as Appendix 4.9A.

Table 4.9.1. Partial List of Water Quality Management Techniques for Reservoirs and Tailwaters

Technique	Application
Dilution and flushing	Improve inflow quality (e.g., reduce nutrient, sediments, and contaminant loads)
Diversion	Improve inflow quality (e.g., reduce nutrient, sediments, and contaminant loads)
Biomanipulation	Reduce algal blooms
Nutrient inactivation	Reduce algal blooms and nutrient export
Nutrient addition	Modify Nitrogen:Phosphorus ratio for algal control
Sediment removal	Remove contaminants and nutrients, increase storage
Sediment oxidation	Inactivate nutrients
Sediment covering	Isolate contaminants, reduce nutrient cycling
Macrophyte harvesting	Remove macrophytes
Herbicide application	Control macrophytes
Algicide application	Control algae

4.9.1 NUTRIENT MANAGEMENT

Nutrient management can be accomplished by management of external sources (inflows, groundwater, surface runoff) and internal sources (mobilization from sediments and plants). External sources are best managed in the watershed at point and nonpoint sources as discussed previously. Often watershed management is not readily available and alternate techniques such as dilution, flushing, and diversion are used. Dilution and flushing is a function of water management and may be accomplished in multi reservoir systems if project operations include some latitude in applying area-capacity curves. The concept here is to pass water with elevated nutrient concentrations from one watershed to another further downstream if you can dilute or flush nutrients with the water management. Diversion is most often applicable to the rerouting of a point source effluent (e.g., sewage outfall) to a detention pond, overland treatment, or larger reservoir.

Physicochemical treatments for in-lake water quality management (excluding mixers, aerators, and oxygenation devices, etc.) include the addition of chemicals for algal and macrophyte control, removal of macrophytes with mechanical harvesting, nutrient inactivation (e.g., phosphorus precipitation with alum), nutrient addition (e.g., addition of nitrate to increase the N:P ratio), sediment covering, sediment removal, and sediment oxidation. With the exception of chemical applications for macrophyte and algal control, these techniques focus on control of internal sources of nutrients (even macrophyte and algal control impacts internal nutrient cycling). The concept here is to limit production by limiting essential nutrients (e.g., phosphorus). Nutrient inactivation and nutrient addition are designed to balance the nitrogen to phosphorus ratio at a value that favors the growth of less obnoxious algal species. Sediment covering, removal, and oxidation techniques attempt to limit available phosphorus from the sediments and water column, thereby limiting algal production. These techniques may be considered as a "bottom up" approach to nutrient management.

4.9.2 BIOLOGICAL MANIPULATION

Biological manipulation, a term first used by Shapiro et al. (1975), involves techniques to manipulate the biological structure in the lake to improve water quality and centers around nutrient availability and the trophic structure or food web of the lake. Usually, the approach is applied to lessen or eliminate nuisance blue-green algal populations, change the fisheries to a more desirable population (e.g., increase numbers or change species composition), or alter the macrophyte community for fisheries or recreation. Biological manipulation utilizes techniques for manipulation of selected levels in the food web (e.g., removal or control of fish that heavily graze on large zooplankton (which in turn remove algae)) or a "top down" approach (Figure 4.9.1). Major factors to consider about the ability of biomanipulation to provide water quality enhancement revolve around (1) the potential to reduce algal biomass where loads cannot be controlled and (2) the potential to augment or accelerate the effects of load reductions (National Research Council 1991). Additional information is available in Carpenter (1988).

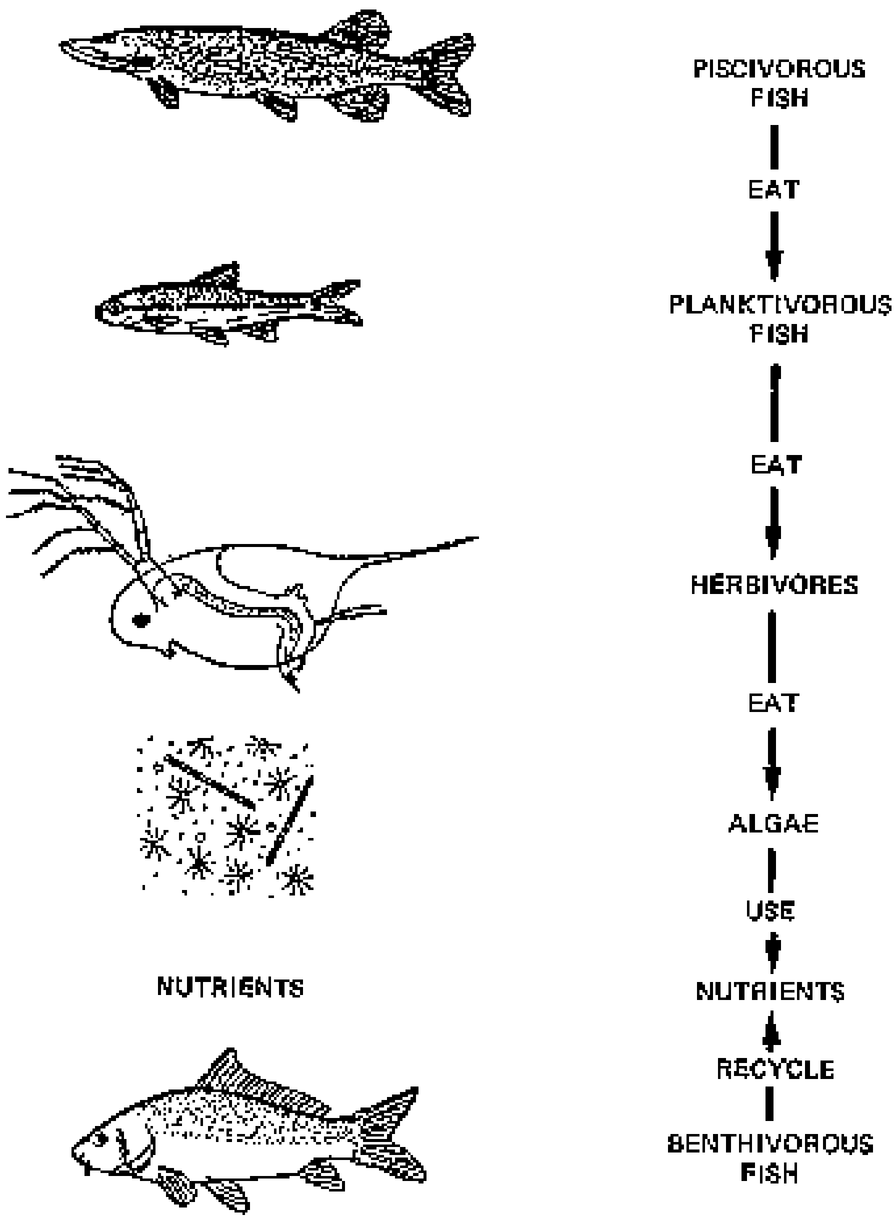


Figure 4.9.1 The aquatic food chain, indicating interactions between the components of the biomanipulation model (after Shapiro et al. 1982)

4.9.3 REFERENCES

Carpenter, S. R., ed. 1988. *Complex Interactions in Lake Communities*. Springer Verlag, New York.

Cooke, G. D. and Kennedy, R. H. 1989. "Water quality management for reservoirs and tailwaters, Report 1, In-reservoir water quality management," Technical Report E-89-1, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

National Research Council. 1992. *Restoration of Aquatic Ecosystems: Science, Technology, and Public Policy*. National Academy of Sciences, Washington, D.C.

Shapiro, J., et al. 1982. "Experiments and Experiences in Biomanipulation," Interim Report No. 19, Limnological Research Center, University of Minnesota, Minneapolis, MN.

Shapiro, J., La Marra, V., and Lynch, M. 1975. "Biomanipulation: An Ecosystem Approach to Lake Restoration," in Water Quality Management Through Biological Control, P. L. Brezonik and J.L. Fox, eds., Department of Environmental Engineering and Sciences, University of Florida, Gainseville, FL.

CHAPTER 5

POST-PROJECT EVALUATION PROCESSES

5.1 ITERATIVE PROCESS OF EVALUATING APPLIED TECHNIQUES

INTRODUCTION

This session will provide an expanded outline that can be adapted to most situations where an enhancement technique has been implemented. The process of evaluating an applied technique should focus on 1) the effectiveness and efficiency of the technique at meeting management objectives, 2) technology transfer of the technique for other applications, and 3) impacts to the ecosystem beyond the water quality problems specifically addressed by the technique (but hopefully considered in the determination of the technique to apply). This process is also described in general terms and will be more fully developed with input from the workshop participants. Workshop participants should receive enough information to adapt and help develop an appropriate evaluation process for any enhancement technique applied in their region. Questions suggested by the National Research Council (U.S.) (1992) for evaluating post-restoration of an aquatic ecosystem may be adapted and used in evaluating the application of an enhancement technique. For example:

1. To what extent were management objectives achieved?
2. Is the system perceived to be or quantitatively better than before?
3. What are the maintenance requirements or is the system self-sustaining?
4. What lessons have been learned?
5. Have the techniques and information gained been transferred effectively to others?
6. What was the time and cost?
7. Were benefits identified, credited, and compared to costs?
8. Were other approaches more applicable?

Topics to be discussed include monitoring, identification of short and long term trends, operation and maintenance, and system optimization.

5.1.1 MONITORING

Monitoring should be conducted as part of the assessment process and should provide a basis for evaluation of post-project conditions if adequate monitoring is conducted upon implementation of the enhancement technique. Monitoring should be specific for the constituents of interest, well-designed and implemented, and periodically reviewed for soundness and application of the data for assessments. Monitoring can be designed to describe both short and long term trends in water quality to determine the effectiveness of the applied enhancement technique and can also provide information on the performance of the technique so that costs may be reduced through optimization of the operation of the system.

5.1.2 SHORT AND LONG TERM TRENDS

Monitoring at an appropriate level (in both time and space) to identify short term trends (diurnal impacts, critical periods, operations) and long term trends (seasonal or annual responses, community structure, trophic status) provides information about the effect of the technique on the ecosystem (watershed, reservoir, tailwater) and may provide information applicable to other sites.

5.1.3 OPERATION AND MAINTENANCE

Operation and maintenance of the system implemented should be evaluated and tracked to provide information on performance and costs. Upgrading equipment should be considered as new technology is developed. Capital expenses may be offset by equipment with a longer life or lower maintenance costs.

5.1.4 SYSTEM OPTIMIZATION

Using information from monitoring and records of operation and maintenance, the application of a technique may be optimized to improve efficiency and reduce cost. Knowledge of temporal and spatial processes may also result in reduced operation and a decreased operating cost. Using the system in conjunction with other techniques or at various levels will also decrease costs.

5.1.5 REFERENCES

National Research Council (U.S.). 1992. Restoration of Aquatic Ecosystems, Science, Technology, and Public Policy, Committee on Restoration of Aquatic Ecosystems: Science, Technology, and Public Policy, Water Science and Technology Board, Commission on Geosciences, Environment, and Resources, National academy Press, Washington, DC.